Centre for
Environmental Assessment
of Products and Material Systems

A systematic approach to environmental priority strategies in product development (EPS). Version 2000 – General system characteristics

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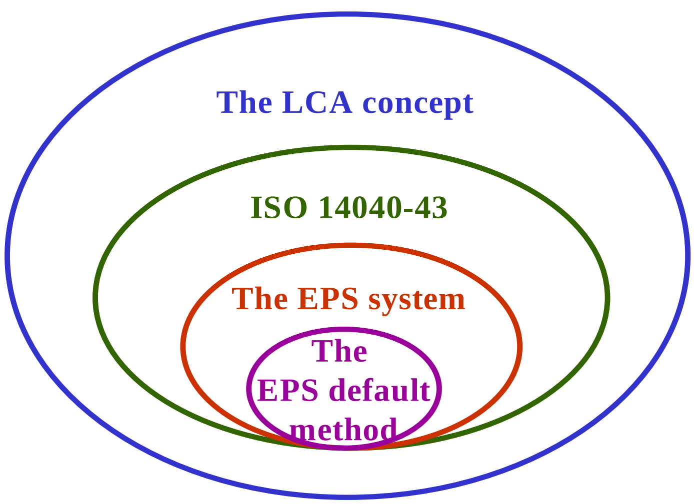

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Foreword

The present version of the EPS system has been developed within CPM (Centre for the environmental assessment of Products and Material systems) a joint research environment at Chalmers University of Technology with participation from industry. CPM is supported by The Swedish National Board for Technical and Industrial Development. Bengt Steen from the department of Technical Environmental Planning at Chalmers University of Technology has been project leader.

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The report is describing the general principles and default methodology of the EPS system. The system principles and methodology is based on earlier versions of the EPS system, in particular the version 1996. The present version has been given the number 2000. In comparison with the 1996 version the basic principles are the same, but the description is more detailed and the ISO standard language is adopted.

Acknowledgement

The EPS system, as it is presented in this report, has taken its form through contributions from many individuals and organisations. It has been a privilege to further the achievements from the Swedish Product Ecology project and much valuable input have come from the co-authors of the EPS system report of that project (Ryding et.al. 1995).

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Content

Summary

This report describes the EPS default methodology and the main principles of the EPS system. The present version of the EPS method is an update of the 1996 version (Steen 1996) and of the 1994 version (Ryding et al., 1995).

Compared to the 1996 version, the EPS system principles are the same. The EPS default method is essentially the same, but, since ISO standards on LCA now are available, it is described in ISO terms (ISO 14040 – 14043). In addition the description is more detailed, and the database updated and extended. To some degree this is in line with requirements put forth by ISO and to some degree this is due to experiences made in the use of the EPS system.

Although the main principles are the same, the elaboration of the descriptions means a new development of the system. As the EPS system becomes more and more developed and spread, the identity of the system shifts from being an informal concept of its creators to be defined by its documentation.

As in many complex systems conflicts often arise between different principles and requirements. The EPS system has an outspoken hierarchy among its principles and rules to help in these situations. This hierarchy is based on a top-down principle giving highest priority to the usefulness of the system.

The other subordinate main principles and characteristics of the of the EPS system are (in hierarchical order):

- The 'index' principle, requiring ready-made indices for materials and processes representing weighted and aggregated impacts
- The default principle, requiring an operative method as default
- The uncertainty principle, requiring the uncertainty of input data to be estimated
- Choice of default data and models to determine them

The choice of default data and models to determine them is by far the most work-intensive part of the system development. This report covers the system principles and general methodology. Default data and specific models are presented in a separate report.

1 Introduction

The need for a better environment is generally accepted in society and numerous activities have evolved with the intention of promoting a sustainable development. The 'Agenda 21' -influenced activities of governments and authorities and the environmental management activities of companies, standardised in the ISO-14000-series, demonstrates this.

Looking back at what has been done so far of the intentions expressed at the Rio conference, you find that there has been an intensive development of management systems.

It is however clear that no management system will do all the jobs necessary for a sustainable development. We need operative tools to decide whether an activity results in an increased or decreased sustainability. For example: a bucket may be made of several types of materials, such as galvanised steel, stainless steel, polypropylene or wood. Which one is to prefer from an environmental standpoint? To answer this question you need not only general guidelines as expressed in the ISO 14000 standards, but also real data and rules how to transform various types of information to a ranking of alternatives.

With a few exceptions operative tools for ranking of alternatives have been designed for the development of ‘green’ products (i.e. where the environmental performance is the key value of the product) or for assessments of existing products. In both these cases the time available for analysis is in the order of months. The Life Cycle Assessment (LCA) methodology was developed in such contexts in during the 80ies and 90ies.

The tool described here, the EPS system, (EPS stands for Environmental Priority Strategies in product design) was developed to meet the requirements of an everyday product development process, where the environmental concern is just one among several others. The development of the EPS system was started during 1989 on a request from Volvo and as a co-operation between Volvo, the Swedish Environmental Research Institute (IVL) and the Swedish Federation of Industries. Since then it has been modified several times during projects, which have involved several companies, like in the Swedish Product Ecology Project (Ryding et. al 1995) and the Nordic NEP project (Steen et.al, 1996). The last modification is made within the Centre for Environmental Assessment of Products and Material Systems, CPM (http://www.cpm.chalmers.se).

The product development process is often regarded as a systematic process and there are several methods described in the literature. However, the reality offers several unexpected events, and substantial changes of plans are common. In the beginning the degree of freedom is high and the cost for changes low. As the process proceeds the degree of freedom decreases and the cost for changes increases (fig 1.1). A tool thus has to be fast in the beginning of the process and cover as many environmental aspects as possible, while it gradually must be able to allow a more detailed and diversified analysis at later stages.

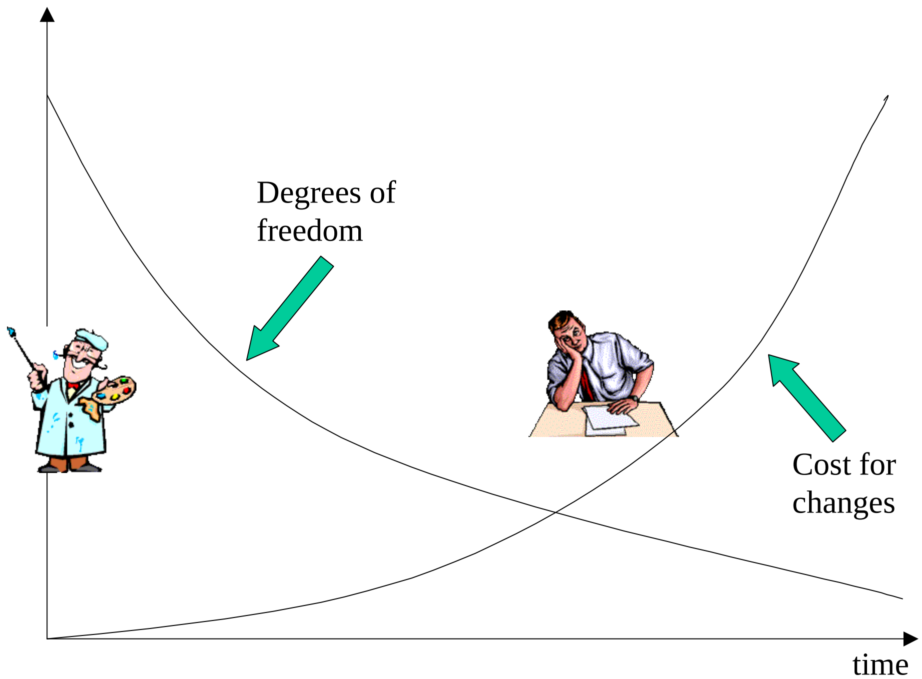

Figure 1.1 The product development process

As a consequence of the requirement to adopt to an everyday product development environment, there is a demand on the use of a language that is easy to understand without too much training. Otherwise the risk of making a tool that may be good, but not used, is high.

2 Goal

Considering the requirements and expectations mentioned in the introduction, a goal was formulated for the EPS system.

2.1 Goal

To be operative in a normal product-developing environment and to be able to assess which of two (or more) concepts that has the least impact on the environment. This means that the system must quickly be able to give recommendations in the early phases of the product development on the basis of general information. During later phases it shall allow more elaborate and precise recommendations and investigations as more detailed and specific information on the concepts become available.
The demand about the system being operative contains a demand on usefulness and cost effectiveness. The extra efforts the designer makes are to result in a reasonable improvement for the environment and the product.

To assess the added value from all types of impacts. This requirement is partly a consequence of the demand on the system to be operative. It is considered unrealistic to take for granted that a product developer, who already has many technical and economical considerations to make, would be able to handle several different impact numbers. He or she ought to have the possibility of choosing the degree of complexity and detail in the information.

To communicate an understanding of the magnitude of the impact. The result of the EPS analysis should be possible to be weighed against other demands on the product.

To offer a forum for growth of a product related environmental strategy within a company in terms of "the 4 p's": plan, pattern, position and perspective. A plan is the original meaning of a strategy. A pattern means that it is not all decided from the beginning. A strategy grows as a pattern from various actions taken develops and many actors contribute. Position means that a company's environmental activities are profiled in relation to its market and competitors. Perspective means that it offers a way of learning.

When formulating a goal based on general information about a need and what might be possible to achieve, you make a choice. There are other options for the goal formulations. The choice of goal may be more restrictive, for instance the weighting against other non-environmental product properties is not normally subject to consideration in LCA. However, this is the reality for a product developer. In one way or another, it has to be decided how far to go in environmental improvement.

There is an ambition that the EPS system shall be able to give an answer within the order of 5 minutes in the first phases of the product development process and still meet all the requirements on transparency given by the ISO 14040-series. As the development process proceeds and the knowledge increases about whom that might deliver materials and which processes to use in manufacturing, the system must allow the exchange of more general data with more specific. A sensitivity analysis must be possible to make in order to guide the user of the system where to improve the product or the data used for the LCA.

The choice of 5 minutes as a target is based on experiences on what might be needed in a very early stage of the product development process. At that stage, several ideas may be tested and one or two may be chosen for further evaluation. 5 minutes is what it takes to describe the materials and processes of a life cycle of a concept that is not too complicated, like a hammer or a torch. Of course, 5 minutes will not do for a whole car or refrigerator, unless you have made a study before and know what the important parameters are. However, most complicated products are made up of components, and for these, the 5-minute target could be relevant. The time necessary for computation and data collection can be seen as negligible in this context, if adequate software and databases are available.

2.2 Scope

The EPS system is mainly aimed to be a tool for a company's internal product development process. It may be used externally and for other purposes, like for environmental declarations, for purchasing decisions, for education or for environmental accounting, but in those cases, the knowledge of the EPS system and its features and limitations is crucial.

The justification of many of the models used in impact assessments and for estimating inventory data relies on the fact that we analyse product systems. Such systems generally contain many emission or resource depletion events in various places, and we can get a fairly good estimate of the added impacts despite not knowing the individual impacts. Like for an aeroplane, the added weight of its next unknown 200 passengers may be estimated with higher relative precision than the weight of its next unknown passenger.

The models used may therefore not be applicable in other contexts. In particular, care should be taken when using the default models and data given in this work for specific impact assessment cases, like single plants or events.

The EPS system is a strategic tool. Like all LCA's its impact assessment is made in relation to a functional unit. This means that there is no possibility of detecting a violation of an emission or a media quality standard. This has to be done with other methods.

2.3 Vision

In common language there is a concept of environmental 'friendliness'. By experts this term is considered misleading as everything we do seems to have more or less negative impacts on the environment. However there is a kind of thinking of an overall impact on the environment that can be used for comparison. In ISO-terms this would be considered to be a weighed result including all types if impact categories.

When the development of the EPS system started, the designers at Volvo argued that they had several thousand decisions to make each year, and that thousands of persons could be involved. It was therefore desirable to adopt the everyday language and thinking of designers. The flow charts used for mass and energy balances by many LCA practitioners at that time might be familiar to chemical engineers, but not to mechanical engineers, who prefer to think in terms of materials and processes.

The designers therefore outlined a calculation process with indices expressing the overall environmental impact caused by a specific amount of materials or processes. As an example a case with a bucket was given.

Suppose a bucket would be manufactured from polypropylene (PP) using an injection moulding process. We would need 2.7 kg of PP. For the PP there is an environmental impact value of say 1.2 ELU/kg representing the overall impact value for the manufacturing of PP. (A cradle to gate process). ELU stands for Environmental Load Units. For the blow moulding process there is an index of say 0.2 ELU/kg. The total environmental impact value for the manufacturing of the bucket would thus be 2.7 kg*1.2 ELU/kg + 2.7 kg*0.2 ELU/kg = 3.78 ELU.
Suppose then that the bucket when it is weird out is deposited in a landfill and an additional impact on the environment occurs. An index of say 0.1 ELU/kg would the give an additional impact of 0.27 ELU.
Other indices could be developed for processes related to PP, like material recycling, energy recovery, incineration etc.

A similar analysis could then be made for a steel bucket to see which life cycle impact were the lowest.

3 EPS system principles

Systems may be described by a set of rules and definitions. The rules of the EPS-system are presented below as 'principles'. The definitions are in agreement with those of the ISO 14040 series. Some terms, which are not used by ISO, are defined in the text.

The development of the EPS system is made in a top-down manner. Starting with the requirements expressed in the formulation of the goal, various methods is developed to produce the data and indices needed for the analysis. In order to make the system operative a default method including a database is developed. The default method is given a version number and is updated at some interval as the knowledge of environmental impacts grows and as current technology changes. The default database could be used in the beginning of the product development phase and the indices gradually exchanged as more specific knowledge of material and processes used develop.

The top-down development of the EPS system leads to a type of hierarchy among the principles and methods used. The top one is for instance demanding ready-made indices representing the total environmental impact and is the most rigid. The bottom ones, like default methods for calculating emissions from various materials in a waste incineration, may be altered as soon as better information is given or in the next version of the system.

Below in this section, the general principles of the EPS system are described in 'hierarchical order'. The intention has been to have as few general principles as possible and as consistently applied as possible. In complex systems the understanding of the results and the possibility of communicating it increase if the system rules are simple. The possibility of understanding the results was one of the major goals for the system development. (See 2.1)

3.1 The top-down principle

When developing a complex system like the EPS system, there will always be parts that are not known or not possible to include because of limited resources for the analysis. Issues in the system must always be dealt with in an economical way. Important issues for the decision(s) at hand are given attention first, and less important issues have to wait. Working in a top down manner is therefore a leading principle. This means two things:

1. issues close to the decision are dealt with before those giving the basic information
2. rough estimates are made first. The quality are improved if experience from sensitivity analysis of real cases have indicated that it is meaningful compared to other issues

3.2 The 'index' principle

The user of the EPS system shall be able to describe a product life cycle in terms of materials and processes for which ready made weighted impacts assessments shall be available in the form of indices. The indices shall represent the weighted and aggregated environmental impact of the production, processing and waste management of materials.

The LCA made by the EPS system as compared to conventional LCA's is shown in figure 3.1

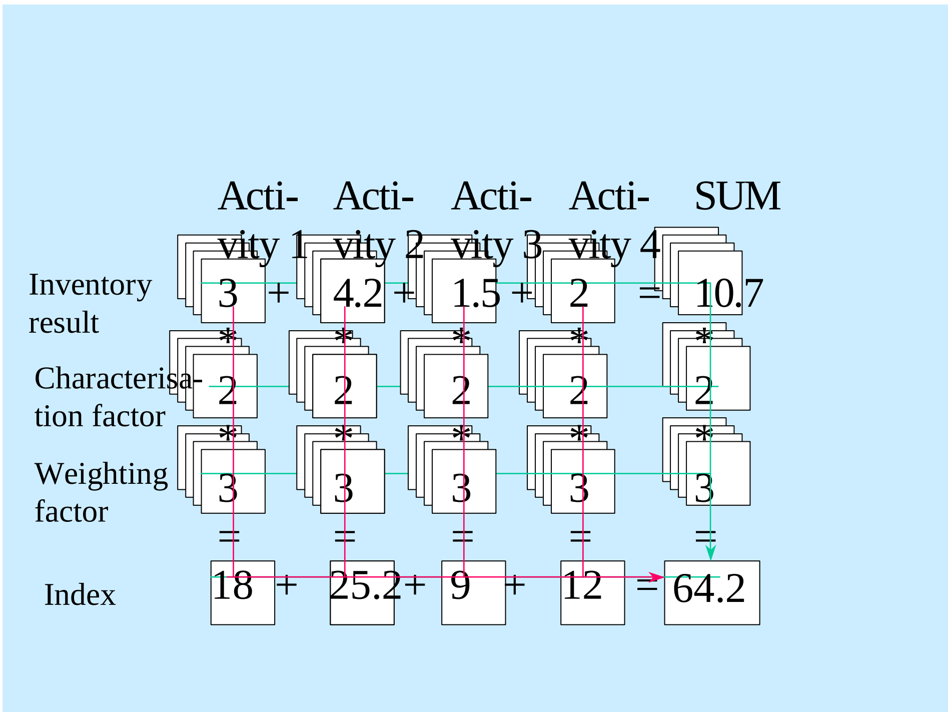

Figure 3.1 LCA by EPS (first vertical and then horizontal) compared to conventional LCA (first horizontal and then vertical)

There has been a debate going on for many years, where several authors express their dissent of the 'one number concept' fearing that the transparency will be lost when an environmental impact is described in one number, like in an index. On the other side, designers often express their need for practical tools that may be used in their everyday life.

This debate is not motivated by a real methodological dilemma. There is no problem of supplying all the information needed for the one number calculation for those who want to see it, especially not if the calculations are made by computers. The problem is more of a communication type. When people not directly involved in an LCA and not being LCA experts are going to cite the results of an analysis they often leave out the background information as it means little to them and as it is time-consuming to render. If there is a 'one number' available, this is the easiest one to report.

Hopefully, standardisation and education will decrease these problems, but probably not eliminate them. A common way of counteracting misuses of ‘one number’ results is to use several types of weighting methods or ‘one numbers’ when communicating with people that are not LCA experts or informed about the limitations and underlying meaning of the ‘one number’.

Holistic approaches are recognised in that they consider the parts in relation to the whole. The 'whole' in the EPS system is the total 'environmental impact load' of the product life cycle. The total environmental load or impact is expressed in ELU, environmental load units. In line with the holistic way, we thus prefer to express all parts in ELU too, so that any decision of adding a material or selecting a waste management process can be directly evaluated versus the whole.

To some extent one may see an analogy in the way an ink jet writer operates. In every moment it produces only a spot on a paper, a ‘one number’ in terms of its position. The information we get comes mainly from the dynamics of the spot, making patterns on the paper. What we see is however not the background processes as such. We see their impact on the ink jet position on the paper. In the same way the information in the ‘one number’ ELU-value lies not in the value itself, but in how it changes when the model input data changes and how it relates to other numbers.

Sometimes the demand for holism come into conflict with other demands, like quality demands. For instance, when an environmental impact is detected, but its quantitative significance is not subject to scientific consensus, the holistic principle is superior and the impact included in the analysis on the basis of whether it is considered to improve the holistic picture or not. Excluding it would be equal to postulating that the extension of the impact was zero.

In natural science the demand for 'true' and robust impact models is much stricter. This does not mean that the EPS system rejects a scientific background, on the contrary. It only means that its focus is on the product performance and not on the impact model's performance. One may say that it has its focus in engineering science rather than in natural science.

3.3 The default principle

The use of default settings in software is common. For instance in Microsoft Word, there are 'wizards', helping you to create letters and faxes. When you use these there are a few options to choose between. One of these options is already marked, and if you just go on with the process of creating a letter or fax format, this 'default' setting will be the one you get. In the beginning, you may just want to write a letter, and the format does not matter. Later on you may want your own style and consequently may choose another alternative.

Using a default approach on the EPS system design is a way of handling the conflict of quickly having to come up with one recommendation of which product alternative to prefer and the realisation of the fact that there may be several answers.

The default approach has an advantage in three other respects.

1. It fits well with the typical progress of the product development process. In the beginning many product alternatives may be considered and a vague idea exists of which deliverers of materials and process equipment that may be contracted. Later on, more focus on a few alternatives and more specific information may be available.

2. The default setting can communicate a company's environmental policy to the designers. The value-laden choices made represent its policy.

3. The analytical process will be faster. Instead of performing an LCA in all possible alternative ways and then drawing the conclusions, the default method is used and depending on the results of a sensitivity analysis alternative options are explored.

3.4 The uncertainty principle

In LCA in general, and life cycle impact assessment in particular, large uncertainties are involved. Mostly, the location of an emission is unknown and hence the effects may not be estimated without great uncertainty. Sometimes emission factors are used when there is a lack of site specific data.

The large uncertainties are a reality and must be addressed in some way. Many LCA practitioners recommend the use of the term “potential effects”. This term is also used by ISO in the standards 14040 and 14042 to indicate that there is an unclear relation between the outcome of a life cycle impact assessment and real impacts on the environment. However this approach does not say anything about what potential means in quantitative terms and the user of the results are left with a warning he or she cannot easily interpret. In the EPS system, an 'uncertainty principle' was adopted at an early stage, saying that any data used in the analysis should be accompanied of a quantitative estimate of the uncertainty.

Again, this principle may cause a conflict with a demand for true and accurate data. There is seldom any accurate or 'scientific' estimation available on data uncertainty. Still, making the uncertainty principle superior, a rough guess of an uncertainty of, say a factor of 10, is more valuable for the overall analysis than just forgetting about uncertainty and acting as if there is no one.

Having estimated uncertainty of input data, the uncertainty of the calculated values may be determined. The methodology used in the EPS-system is described in chapter 4.7.

Product development means choices between alternatives. Often this is possible without precise knowledge about the alternative impacts or impact values. (figure 3.2)

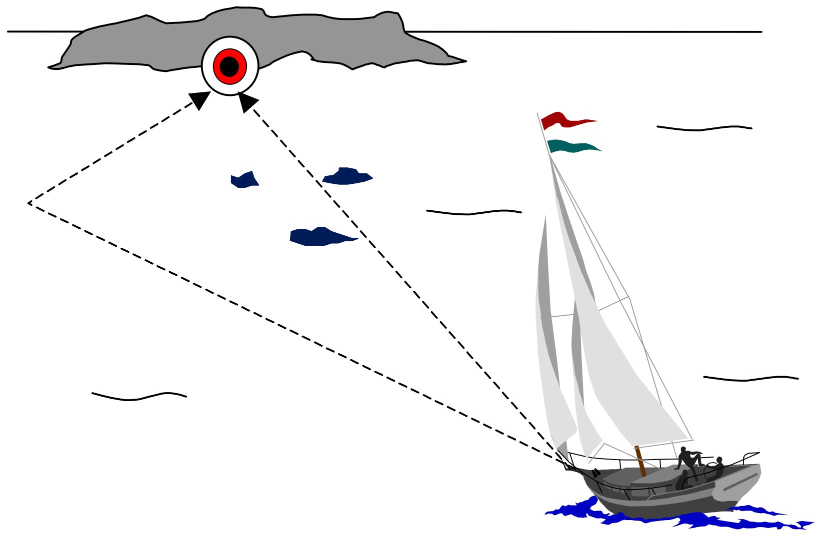

Figure 3.2. When choosing between two alternatives you do not have to know the exact consequences of the alternative that you do not choose.

3.5 Choice of default indices

To make an index, most elements in a full LCA have to be included: the goal formulation, the inventory, the selection of impact categories, the assignment of emissions and resources to impact categories, the characterisation and the weighting.

Before looking at these elements, which will be done in chapter 4 below, a few more general system principles will be lined out:
1) a default approach in terms of environmental philosophy,
2) a 'causality principle' and
3) a 'precautionary principle'.

3.5.1 Environmental philosophy

In order to be able to find a measure of the environmental impact that could be transformed to an index, we need first to answer two questions: when does the environment improve and how can we determine its value or change in value?

We need thus a reference of some sort and a way of weighing deviations from the reference.

Today virtually all methods for weighting impacts across impact categories either use environmental goals or the present situation as a reference.

Environmental goals may be formulated in various ways: from very general attitudes and wishes concerning life qualities to local operative goals for emissions or recycling numbers.

On a general level, goals relate to environmental philosophy or environmental ethics. This is a discipline in itself and there is an extensive literature on the subject e.g. Shrader-Frechette (1991). Human approaches to the environment are often described in various grades of anthropocentricity or biocentricity. Beltrani (1997) discusses ethical approaches in relation to LCA and choice of safe guard subjects.

Hofstetter, (1998) has applied cultural theory to classify people in their approaches as fatalists, individualists, egalitarians, and hierarcists and used this to describe various weighting principles.

On an individual level one may also describe approaches towards the environment or towards other humans and other species in terms of empathic capacity. If a person is rich - in a wide sense – he or she can afford to care more for others than else. If a person is poor, he or she cannot afford to care for others and behaves more egoistic.

In a way the economy for a person represents an egoistic perspective. When choosing a default index method there would not be much of an extra information if the indices represented such an approach. To maximise the 'resolution' of the information given to a designer, the traditional economic perspective ought to be complemented with a measure representing as much of an empathic capacity as possible without abandoning accepted attitudes amongst people, i.e. loose its relevance. In that way the indices would get a maximum information value together with the economic information.

An approach representing a high degree of concern with other individuals and species, but not leaving the anthropocentric perspective is found in the environmental goals set up by the “Earth's Summit” at Rio de Janeiro 1992.

In the development of the EPS system, it was decided to choose a default evaluation of environmental impacts which as much as possible was compatible with the goals set by the earth summit at Rio.

The Rio conference is to a large extent evaluating the environmental impact in terms of its relation to a "sustainable development". This means that the interest of resources increase compared the focus of earlier environmental concern: effects of emissions. The Rio conference deal with resource aspects not only in terms of natural resources but also in terms of society's ability to respond to environmental threats.

It has not been possible to find a measure of society's ability to adjust to environmental threats and in particular to determine how this is influenced by a product concept. Therefore this particular aspect is left out of consideration in the EPS system.

The Rio protocol was not the result of an isolated event. It was to a large extent reflecting the current attitudes on environmental issues, let be of governement 'environmentalists'. The issues brought up in Rio were issues that had been under discussion for many years, issues that you may find in most comprehensive literature on environment and in national environmental goals.

At the Swedish Environmental Protection Agency, environmental issues are described as 'threats' and 'safeguard subjects'. Threats are mechanisms, like acidification and global warming. Safeguard subjects are the things we want to safeguard in the environment, like human health and bio-diversity.

The EPS default method evaluates impact on the environment via its impact on one or several safeguard subjects. These have been chosen from those that were included in the Rio protocol, although not necessarily explicitly formulated there: human health, resources, ecosystem production capacity, bio-diversity and esthetical values. Today the safe guard subject ‘esthetical values’ is extended and named ‘cultural and recreational’ values and resources are specified as ‘abiotic stock resources’.

You may argue that bio-diversity and ecosystem production capacities are resources or that everything is of interest because it sooner or later impacts on human health. But if you do not know how certain threats will develop, uncertainty may call for a separate guard.

Now, if the default environmental goal used in the EPS system is chosen to be the preservation of the safeguard subjects, a reference state has to be chosen and a way of weighing deviations from the reference state. In line with the goal to produce an understandable answer and to have as few rules as possible, the simplest solution to the reference problem is to chose the current state of the safeguard subjects, and only look for changes in the safeguard subjects. The present state ought to the one that is easiest to describe. Besides, in practical use of LCA tools we can hardly ever use anything else than linear relationships. Under those circumstances, the choice of reference state will not influence the result, at least not in an analysis of incremental environmental changes caused by human activities.

How do we then weight various changes in the safeguard subjects towards each other?

In the goals and superior principles given above there are some requirements that influence the choice of weighting principle. First we have the demand on an understandable measure. Second we have an orientation towards sustainability in our environmental philosophy. Sustainability has very much to do with resources and reserves. On a long-term basis it is more or less impossible to foresee all problems that will occur. A good strategy is to keep resources to be able to solve the problems.

Therefore a monetary approach is chosen.

An interesting parallel may be found in psychotherapeutic strategy. If increasing the mental capacity of the patients, they are able to solve the various problems they might face to the best (Pedersen, 1986). This strategy may be compared to the traditional treatment of physicians: to eliminate the problem at hand. Lohman (1969) concludes that in health care, problem elimination is a dominating activity. He explains this with the enormous impact Pasteur and his successors have had on the society and on disciplines outside their own. Pasteur showed that it was possible to find the evil and cure it.

In environmental strategies the ‘problem eliminating strategy’ is dominating and has been for long. Among LCA experts there is a common way of expressing the environmental goal as "less is better" If there are no emissions or resources used, the environment will be OK.

Looking at some distance at these two strategies, it seems reasonable that the problem eliminating strategy is applied in acute situations and in a short time perspective. For longer planning, a more resource-oriented approach is to prefer.

A resource oriented, widely understood measure is the monetary measure. However this can be expressed in several ways. In the EPS system a kind of ‘willingness to pay’ (WTP) to restore changes in the safe guard subjects have been chosen as the monetary measure. The WTP is measured in today’s OECD population and applied to all those, who are affected by a change. No discounting for future effects are made as future generations have the same right to a good environment as we have (Rio Convention). The basic values of the environment are not considered subject to change. The OECD values of today are used even for impacts on people outside OECD and for future generations. This way of looking at the impacts may be called anthropocentric altruism. Willingness to pay is understood as an expression of an attitude in monetary terms towards a change regardless of whom is guilty to the change. The reason for using the OECD values of today for other populations are mainly two: 1) it is practical in that it is measurable, 2) it is mostly the OECD inhabitant of today that are making the decisions as designers.

An alternative had been to choose the restoration cost, or willingness to accept (WTA). Restoration cost would have given some unrealistic results, since we often chose to live with environmental degradation or positive changes rather than restoring them at unreasonable high cost. Bad odour is one example, noise is another, meadows created by grazing cattle’s a third. WTA is more difficult to measure than WTP but given the modifications of WTP expressed above the difference decrease. WTA also has the problem of claims for compensation that are much higher than the available money. This problem is big enough with the WTP approach as it does not ‘cost’ anything to express a high WTP. To some degree, the technique of determining WTP can decrease this problem, but not fully. Some results, like results from CVM studies, (see 4.6.2), are not directly additive in a strict economic sense. If one wants to use CVM-based WTP:s together with WTP determined by other methods, for instance hedonic pricing, one has to ‘translate’ the levels.

Many environmental economists use various discount rates (ExternE, 1995). However, even at very low discount rates effects lasting for hundreds of years may be overlooked. For instance the greenhouse effect tend to be more or less negligible in some studies (Azar,1996).

The WTP as used in the EPS default weighting method is separate from the WTP used in many cost-benefit studies in that is does not include direct impacts on the economy. For instance, a loss of income due to hospitalisation which is included in the ExternE study (1995) is not included in the EPS default weighting method, as the economic system is not included in the safeguard subjects.

The values of the WTP will change from person to person and from generation to generation. This is not a deficiency, it is simply a part of reality. Different experiences and life situations most likely will result in different attitudes to changes in the environment. For future generations we would ideally like to include their attitudes. But it is very difficult to understand what another person will think about changes in their life conditions. The most common way of approaching this problem is the one we teach our children. We would ask them: what would <u>you</u> think if this happened to you?

The WTP as it is used here is not an ultimate WTP. For instance, if there were very little food available, the WTP for crop would probably be as much as there was money available. The WTP, which is chosen in the EPS default method, relates to everyday life conditions. Normally you are not willing to pay more than it takes.

There is an easy way a designer can understand the default indices and the results of an calculation with the indices. They represent the money he or she together with other OECD inhabitants would be willing to pay, to avoid the impacts from the design he/she considers. Another way of looking at the indices and the impact values is as representing an average risk. Risk is normally understood as a probability of an event times a consequence. The indices express the most probable change in the environment times its consequence in terms of WTP.

3.5.2 The causality principle

When looking at results of LCA’s, it is sometimes unclear whether the results represent a consequence of a change or an allocated record of some kind. For example if 11 persons have one litre of waste water to get rid of and pore it into a waste water system having a container with a capacity of 10 litre, there will be a spill of 1 litre on the floor. (fig 3.3)

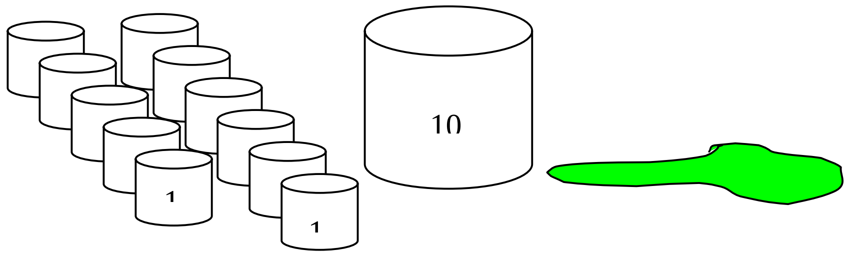

Figure 3.2 Consequence or guilt?

What environmental impact should be allocated to each person? If we think in a guilt perspective, it would probably be 1/11 of a litre, unless they knew that there would be an overflow. During that circumstance we might be reluctant to say that each one who knew that if he or she did not put his/her litre into the waste water system, there would have been one litre less on the floor, would have a guilt of 1 litre. Maybe you can argue that the last person bears most of the guilt, as it was actually he or she that caused the overflow. Anyway, it is obvious that there is no ‘scientific’ answer to the question of how to allocate 'guilt' or 'benefits' unless they are related to consequences.

The EPS system makes it a principle to estimate environmental consequences of various human activities. Even if this sometimes is more difficult in terms of finding indisputable data there is a ‘true’ answer that at least can be conceptualised.

The causality principle demanding assessment of consequences has in its turn some very significant implications for the way an EPS analysis is made and on its results. Some implications make it easier to perform an analysis while others make it more complicated.

For instance the consequence of specific individual 'events’ are damped or levelled out as large systems in terms of markets and infrastructure are involved. This means that average or long term marginal values can be used instead of a multitude of individual data linked to special process conditions which take place when a individual piece of a product were manufactured.

So, when a customer in a shop buys a red cup instead of a blue, much of the environmental impact connected to that cup has already taken place. The factory making red cups will however note that the sales of red cups have increased and will most likely produce another one and purchase some extra raw material and energy from the market causing some extra emissions and resource depletion on the marginal. Because of the causality principle, the best estimates of these emissions and resource flows are thus the average values or average marginal values and not the very specific emissions that occurred when the cup was produced.

A problem with the 'causality principle' is that it is unclear if there is an end to the consequences of a certain human activity. In a way you may change the ‘path of history’ for each activity you carry out. When a factory is built at some location, the birds have to hedge somewhere else and will possibly form other couples and bring up other individuals. In the long run the number of birds may stabilise at a new level, but the change of individuals from a unique line of successors to another unique line of successors will probably never end.

In order to decrease the problem of unforeseeable consequences, we may limit our analysis to some general properties of the environment - impact categories and indicators representing changes of general properties of the safeguard subjects.

A complication when looking for causalities is that the marginal changes may be difficult to estimate if there are non-linear dose-response functions. For instance if there is an S-type dose-response, or emission-response curve for a particular source, we may have a situation like in figure 3.3.

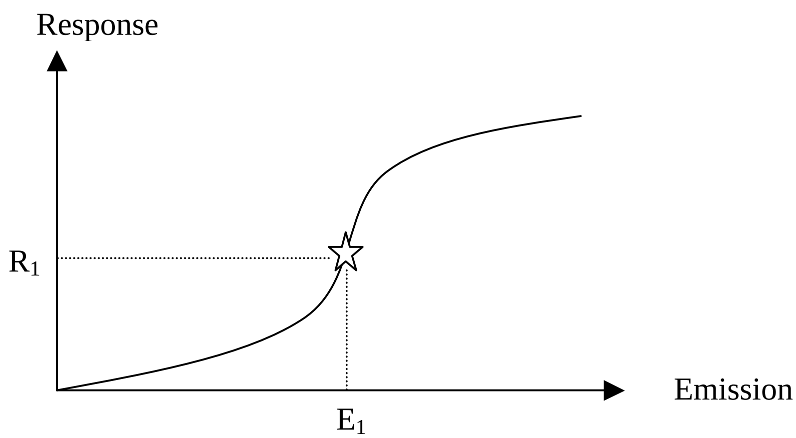

Figure 3.3 S-shaped emission-response curve for a particular source

If an emission, E_1, were like indicated in figure 3.3, there is a relatively large change in response, ΔR per change in emissions, ΔE. $\Delta R/\Delta E$ is large compared to R_1/E_1. If the emission would be more or less, the $\Delta R/\Delta E$ would be less.

If there is number of emission events from the same source there may be a situation, like in figure 3.4.

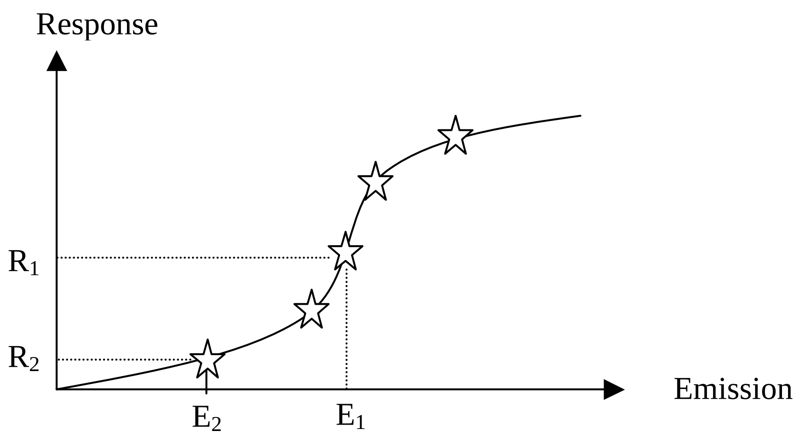

Figure 3.4 Several emission-response events and an S-shaped emission-response curve for a particular source

If a number of emission events are randomly distributed as indicated in figure 3.4, the average $\Delta R/\Delta E$ is approximately equal to R_m/E_m, where R_m and E_m are the average values of R and E respectively.

If the position on the emission-response curve is not known, R_m/E_m is used as a best estimate and the possible variations in $\Delta R/\Delta E$ used to estimate the uncertainty.

3.5.3 The precautionary principle

At the Rio Conference, a precautionary principle was adopted.

For the EPS default method, the precautionary principle is applied in two ways.

First, no future technical solutions are encountered. For instance, when modelling the environmental consequences of CO_2-emissions future emissions are estimated assuming 'business as usual'.

Second, models of negative impacts are used at an early stage in terms of scientific consensus. The models have to be published by competent scientists, but the demand on general acceptance is moderate.

The reason for the first way of using the precautionary principle is in a way trivial. If you want to develop a tool to show how to improve the environmental performance of products, it would not be very effective if it assumed that problems were solved in advance.

The second way of using the precautionary principle is quite common in society. For instance, when TLV-values are set for occupational exposure of chemicals, a safety factor is added for uncertainty. If the toxic mechanisms are well known, like for CO, the safety factor between 'no effect level' and the limit value is only about 2-3, while for toluene which has some diffuse effects on the central nerve system, a factor of more than 100 is used.

However when optimising an overall environmental performance of a product, a too ambitious use of safety margins will not be beneficial to the overall solution. If possible it is better to use the uncertainty principle, which calls for a best estimate and a measure of uncertainty. In the EPS system, this principle is superior to the precautionary principle.

3.6 Alternative indices

As indicated in 3.2 there are numerous ways of making indices or LCA:s. Alternative value choices during the inventory and impacts assessment phases most likely result in different indices. During the development of the EPS system in the early 90-ies it was concluded that there was a great need for a structured database and a documentation format to document all these choices and data in an ordered manner. In the Nordic "NEP-project" a data documentation format named SPINE, (Sustainable Product Information Network for the Environment) was developed (Steen, 1995). SPINE consists of about 30 tables. The SPINE concept is described further on the web-site www.cpm.chalmers.se

In this chapter (3.5), some general principles for alternative indices will be discussed.

When interpreting the sustainability concept in a company, it is relevant to ask: sustainability for whom? Is it for the human race, for the culture, for our generation, for any individual or for the company? Depending how we answer this question our indices and the 'optimum' products will be differently shaped.

Indices may thus be designed to represent environmental goals having different extensions in time and space. For instance the five-year national emission goals may be used for a product with a lifetime of less than five years, which is made and used within the country and which cause only local impacts. In those cases a company may reduce the cost for cleaning equipment by selecting products with low emissions of the substances on the reduction list.

There are several weighting principles based on national reduction goals for emissions, the ‘Ecoscarcity method’ (Ahbe et al.,1990), the ‘Environmental theme method’(Baumann et al. 1992) and the ‘EDIP method’ (Wenzel et al, 1997). Most of them give equal weight to all goals and just weigh according to the relative contribution to achieving the goal. Krozer (1992) has developed a weighting system, which aims at avoiding future costs meet the national emission goals.

In the Tellus method (1992) the weighting is done according to the maximum willingness to pay for cleaning of flue gases.

The Eco-indicator 98 method (Goekoop et al.,1998) and the EPS default method focus on damage or end point effects. The Eco-indicator 98 use a two step weighting procedure, where the first step is made within each safe guard subject (resources, ecosystem health and human health) following a formal methodology. The second step is of panel type.

Several reviews of weighting methods have been performed in the last years (Lindfors et al., 1994), (Lindeijer, 1996) and (Bengtsson, 1998).

Table 3.1 below is partly based on these, but structured a little different and includes only a few of them to illustrate the principles used.

Method name	**Environ-mental goal or reference**	**Weighting principle**	**Spatial extension**	**Type of impact category indicators**
Eco-scarcity	National emissions	Relative reduction of distance to target	Switzerland, Netherlands, Sweden or Norway	Emissions
Eco-Indicator 98	Present state	Two step weighting, last step of panel type	Europe	Damage
EDIP	Present state	Separate weighting of emissions (political goals), resources(supply horizon) and work environment	Global and national	Normalised impact potentials (person-equivalents)
Environ-mental themes	National critical loads	Relative reduction of distance to target	Switzerland, Netherlands, Sweden or Norway	Impact potentials based on chemical, physical or biological properties of emitted substances or resources
EPS-default	Present state of environment	WTP to avoid changes	Global	End point effects
Tellus	Zero emission (not explicitely expressed)	WTP for flue gas cleaning	USA	Emissions

Table 3.1 Different weighting methods

EPS default indices can also be modified for other system borders in time and space. Even site-specific data may be determined. As the product development process proceeds, knowledge about the product system increase, and it is possible to introduce data from specific industrial plants and market regions.

4 The EPS default method

The ISO 14040, 14041, 14042 and 14043 standards are framework standards specifying the necessary elements and steps in performing a standardised LCA. Part of the strength of the LCA concept lies in the communication made possible through simplified models and a harmonised language. There is an advantage in trying to follow the standard as far as possible. In this chapter the EPS system rules and terminology are described in agreement with the ISO framework. However, it is important to remember that the ISO standards were written with respect to specific LCA studies. The EPS system is in itself a framework, although somewhat more specified than the ISO framework. The EPS system contains an even more specified methodology, the EPS default method, which gives a starting point for LCA:s within the EPS system. Figure 4.1, outlines the relation between, the LCA concept, ISO framework, EPS system and the EPS default method.

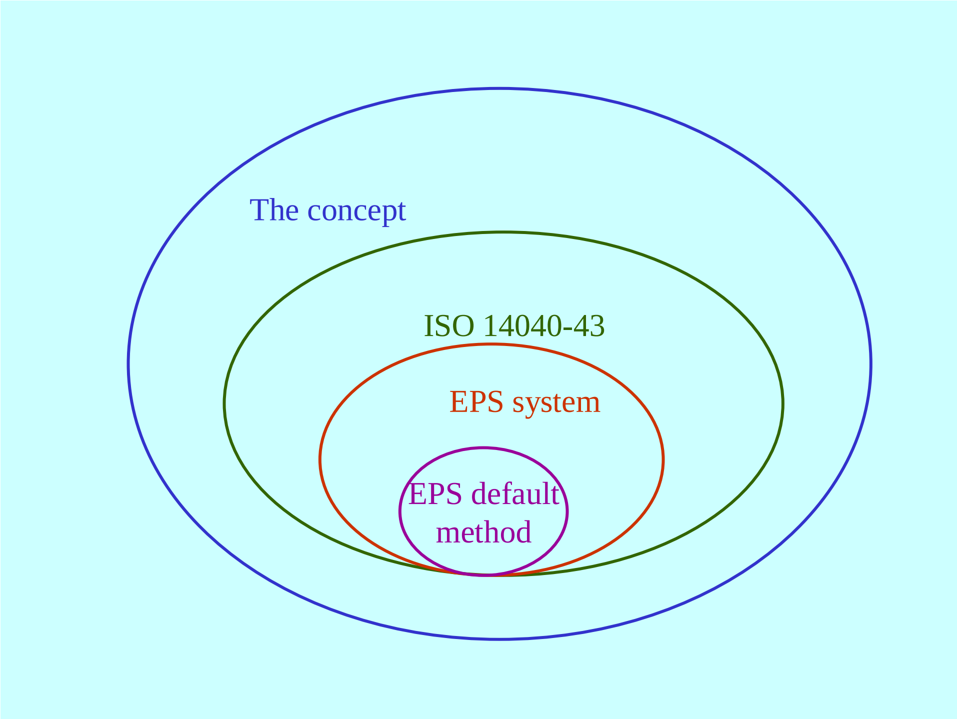

Figure 4.1 Relation between LCA concept, ISO standard framework, EPS system and EPS default method

4.1 Goal and scope

ISO 14040 requires the description of the goal to include the intended application, the reasons for carrying out the study and the intended audience.

The *intended application* is for choosing between design options in product development.

The *reason of carrying out the studies* is to indicate which one of two concepts that are least impacting on the environment.

The *intended audience* is those involved in product development.

In ISO and 14040 there is also list of issues to be covered in the scope formulation of an LCA. This list is repeated and answered in table 4.1.

4.2 Inventory

The inventory required to create an index is by large similar to a conventional life cycle inventory. There is however some characteristic features of the EPS default indices, which will be mentioned here.

One is the development of indices in a 'family' for each material representing various cradle-to gate-, gate to gate- and gate to grave processes. An example of a ‘family of indices is shown in table 4.2.

A second is the focussing on the emissions and resources, which have significant weighting factors and give major contributions to the index values (4.2.1).

A third is the concept of 'structural values' (Karlsson 1995) allocating 'avoided emissions and resource depletions' to materials when they are recycled (4.2.2).

A fourth is a set of 'support methods' for estimating emissions from processes or product systems, where no data are available. These methods are presented in a separate report on models and data (Steen 1999).

Requirement for goal and scope description by ISO 14040	Approach followed in the EPS default method
The functions of the product system(s)	Defined in the specific study
The functional unit	Defined in the specific study
The product system to be studied	Product systems are defined for each study, but a database is required containing subsystem LCA’s of manufacturing, use, processing and waste management of construction materials.
The product system boundaries	Defined in the specific study. The system borders in the sub-systems of the database are: cradle-to-gate, gate-to-gate, manu-facturing-, support- and maintenance processes and gate-to-grave.
Allocation procedures	Similarity to economic system (Costs and values),(4.2).
Types of impact and methodology of impact assessment, and subsequent inter-pretation to be used	Impacts on five safeguards subjects, where impact category indicators are of damage-type and are chosen late in the cause-effect chain (4.3). WTP for avoiding changes are used for weighting.
Data requirements	The designer shall only need to know amount of materials and components, and general data on processes and waste management. The EPS system maintenance staff makes all inventories and modelling of characterisation and weighting factors, which are needed for the database. These data shall consist of a best estimate and a quantitative uncertainty estimate as well as specifications for coverage in time, space and technology. Global data for 1990 is default.
Assumptions	Business as usual is the default scenario for future technology.
Limitations	Is not intended to detect violation of local standards and limits.
Initial data quality requirements	Rough estimates are allowed as long as the uncertainty involved also is estimated.
Type of critical review	Defined in the specific study
Type and format of the report required	Defined in the specific study

Table 4.1 Scope of EPS studies.

Type of process	Material	Index (ELU/kg)
Manufacturing	Polyethylene-LD	1.14
Material recycling	Polyethylene-LD/HQW*)	-0.912
Incineration with energy recovery	Polyethylene-LD/HQW	-0.0115
Incineration	Polyethylene-LD/HQW	0.200
Composting	Polyethylene-LD/HQW	0.200
Landfill	Polyethylene-LD/HQW	0.0782
Lost	Polyethylene-LD/HQW	20.2
Material recycling	Polyethylene-LD/LQW*)	-0.570
Incineration with energy recovery	Polyethylene-LD/LQW	-0.0115
Incineration	Polyethylene-LD/LQW	0.200
Composting	Polyethylene-LD/LQW	0.200
Landfill	Polyethylene-LD/LQW	0.0782
Lost	Polyethylene-LD/LQW	20.2

Table 4.2 Example of a ‘family’ of indices related to a material. *) HQW means ‘high quality waste’, which normally comes from the production process and LQW means ‘low quality waste’.

4.2.1 Focussing on important emissions and resources

Having chosen a weighting principle you may easier focus on those parameters that are important for the result. In conventional LCA without weighting, you cannot exclude any information, and you often see very long lists of inventory parameters which is very time consuming to collect and which have negligible influence on the decisions made. Much of the data on emissions from industrial activities are there because of a monitoring program, the purpose of which is to ensure that no significant environmental effects occur in the vicinity of a plant. There may thus be a negative correlation between data availability and data importance.

Sometimes important inventory data are not available, because it was not measured or reported. When having a default weighting and valuation method, it helps in asking for inventory data of interest. If no data is available, a rough estimate is considered better than just leaving a data gap.

Making rough estimations or guesses is less devastating for the credibility of the study when an estimate of the uncertainty is made together with the best estimate and when a sensitivity and uncertainty analysis is included in the study.

4.2.2 Allocation

When several products or product systems share the same of emissions and resource flows, problems arise on which part to allocate to which system or product flow. This is a fairly complicated issue that hardly can be investigated in detail in a design process.

In order to conceptualise the allocation result for a material or product system, the term ‘structural value’ is used. (Karlsson, 1995). The structural value has a similar meaning as an economic value. The structural value is the weighted, avoided emissions and depleted resources, which come as a result of leaving a material or product to a recycling pool or market. The structural value for a material, which is recycled to be used for the same purpose as the virgin material, is equal to, the weighted emissions and depleted resources of producing the virgin material, minus, the weighted emissions and depleted resources when collecting and restoring the qualities of the virgin material.

Following a top-down procedure, the structural values (primarily in terms of emissions and resource amounts) are first estimated by approximate methods and later analysed for more specific cases. Methods used for approximate determination of structural values are presented in the models and data report. (Steen, 1999).

4.3 Selection of default impact categories and category indicators

Various types of impact indicators can be chosen. At an early stage the criteria in table 4.3 for choice of impact categories was formulated for the EPS system.

Criteria for identification and selection of default impact categories and category indicators of the EPS system
1. The impact categories shall fully cover all significant types of environmental effects due to human activities, without overlapping. 2. The impact categories shall allow a quantitative characterisation of emissions and other human activities in terms of category indicators. 3. The impact categories and indicators shall be possible to understand for laymen. 4. The impact categories shall allow weighting of indicators across categories 5. The impact categories and indicators shall be common to all types of environments. A change of a land area from forest to agriculture should be possible to evaluate

Table 4.3 Criteria for identification and selection of default impact categories and category indicators of the EPS system

In the ISO 14042 standard, the requirements made for the selection of impact categories, category indicators and characterisation models in an LCA study are:

a) the selection of impact categories, category indicators and characterisation models shall be consistent with the goal and scope of the LCA study;

b) the sources for impact categories, category indicators and characterisation models shall be referenced;
c) the selection of impact categories, category indicators and characterisation models shall be justified;
d) accurate and descriptive names shall be provided for the impact categories and category indicators;
e) the selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied, taking the goal and scope into consideration;
f) the environmental mechanism and characterisation model which relate the LCI results and category indicator and provide a basis for characterisation factors shall be described;
g) the appropriateness of the characterisation model used for deriving the category indicator in the context of the goal and scope of the study shall be described.

In addition, the following recommendations is made for the selection of impact categories, category indicators and characterisation models:

a) the impact categories, category indicators, and characterisation models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body;
b) the impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators;
c) value-choices and assumptions made during the selection of impact categories, category indicators, and characterisation models should be minimised;
d) the impact categories, category indicators, and characterisation models should avoid double counting unless required by the goal and scope definition, for example when the study includes both human health and carcinogenicity;
e) the characterisation model for each category indicator should be scientifically and technically valid, and based upon a distinct identifiable environmental mechanism and/or reproducible empirical observation;
f) the category indicators should be environmentally relevant;
g) it should be identified to what extent the characterisation model and the characterisation factors are scientifically and technically valid.

There is a conflict between the requirement of understandable impact indicators and quantitative information (table 4.3). In a cause-effect chain the quantitative relations between an emission and its environmental consequences are better known for the early stages than for the endpoints. For instance the amount of acid produced from an emission of SO_2 can be fairly well modelled, while the reduction in tree growth is more uncertain. However the possibilities to reach consensus about the value of a certain amount of wood is much better than about the value of a certain amount of acid.

For some types of cause-effect chains (or rather: networks) the knowledge is better than for others. For many human health effects there is an extensive literature describing the emissions-dispersion-exposure-dose-response-chain and an end point effect may be described in terms of morbidity or nuisance. For bio-diversity however, it is more difficult to follow the cause-effect chains. Our understanding of the problem is immature in a quantitative sense and it seems necessary to choose impact categories at a relatively early stage in the chain.

An important criterion for choosing impact classes is full coverage of all types of impacts. The types of impacts that should be covered in an LCA are fairly well agreed upon. ISO/DIS 14040 and SETAC code of practice request the same types of effects that were main themes at the earth's summit in Rio: human health, ecosystem health and natural resources.

In the EPS system the impact categories are identified from five safe guard subjects: human health, ecosystem production capacity, abiotic stock resources, bio-diversity and cultural and recreational values.

In earlier versions the third safe guard subject was named 'natural resources' or 'resources', but is now specified as 'abiotic stock resources'. The last safeguard subject was named aesthetic values, but has now been renamed to 'cultural and recreational values' to cover a broader aspect.

4.3.1 Human health impact indicators

A wide spectrum of environmental impacts on human health is described in literature. Some of the impacts have known mechanisms and some is known from epidemiological studies. The epidemiological studies are valuable as they result in quantitative relations and as they use response parameters which are experienced by individuals. This meets the second and third criteria mentioned in table 4.3 for the selection of impact categories. However epidemiological studies do not cover all types of health effects and therefore information from dose-response studies and other studies must be added in order to select category indicators that can fit into characterisation models.

When choosing impact categories, category indicators and characterisation models there are a similar situation as when deciding upon air quality criteria (WHO, 1987). Basic information is compiled in a comprehensive way, but there are many elements of judgement involved to make the final recommendation. For instance, when individuals are exposed to various concentrations of air pollutants and respiratory resistance is measured. The concentration where no effects are measured is determined. Often information is given about how the individuals in a medical sense reacts on elevated concentrations, like increase in respiratory resistance, but generally no description of how the test persons feel are given.

Strand (1991) and ExternE (1995) has reviewed human attitudes towards changes in health conditions in terms of willingness to pay, and from these studies it is possible to see for which types of health category indicators one may find a WTP estimate.

Human health impact indicators may be chosen either to be numerous and very specific, or to be less and more general. The first may be tempting for the medical expert, but the information on corresponding cause-effects available today does not allow the use of diagnostic refinements more than in a few cases. There is also a limitation of studies of attitudes to many environmental related health effects. Besides if there is too many indicators, it will be difficult to obtain a comprehensive view of the indicator results. This together speaks for a limited number of health impact categories. In table 4.4 impact categories and category indicators chosen for the EPS system are shown.

Impact category name	**Category indicator name**	**Indicator unit**	**Notes**
Life expectancy	Years of lost life, (shortname: YOLL)	personyear	Instead of excess mortality, which was used in earlier versions
Severe morbidity and suffering	Severe morbidity	personyear	Including starvation
Morbidity	Morbidity	personyear	Like a cold or flue
Severe nuisance	Severe nuisance	personyear	Would normally cause a reaction to avoid the nuisance
Nuisance	Nuisance	personyear	Irritating, but not causing any direct action

Table 4.4 EPS default impact categories and category indicators for human health effects

It may be relevant to separate morbidity from starvation, as starvation is an important end point for environmental impacts and different in character to normal morbidity, but at present the understanding of what starvation means is poor for laymen in the industrial world.

The choice of impact category and category indicators is evaluated against the ISO 14042 in tables 4.5 and 4.6

ISO requirement	How the choice of EPS default health indicators comply
The selection of impact categories, category indicators and characterisation models shall be consistent with the goal and scope of the LCA study	Allows weighting and communication
The sources for impact categories, category indicators and characterisation models shall be referenced;	Through this report and the models and data report (Steen, 1999)
The selection of impact categories, category indicators and characterisation models shall be justified	Through this report and the models and data report (Steen, 1999)
Accurate and descriptive names shall be provided for the impact categories and category indicators	The names are descrip-tive, but the accuracy is not fully obtained through the name
The selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied, taking the goal and scope into consideration	Meets requirement of including human health

Table 4.5 EPS default indicators compliance with ISO 14042 requirements for the selection of impact categories and category indicators for human health

ISO recommendations	How the choice of EPS default health indicators comply
The impact categories, category indicators, and characterisation models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body	Limited compliance at present stage
The impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators	Compliance because indicators are chosen at the endpoint level
Value-choices and assumptions made during the selection of impact categories, category indicators, and characterisation models should be minimised	The intention is followed, but the compliance is difficult to verify
The impact categories, category indicators, and characterisation models should avoid double counting unless required by the goal and scope definition, for example when the study includes both human health and carcinogenicity	Reasonably well met. There are some risks of double counting with cultural and recreational values.
The category indicators should be environmentally relevant	Documented through this report and the models and data report (Steen, 1999)

Table 4.6 EPS default indicators compliance with ISO 14042 recommendations for the selection of impact categories and category indicators for human health

An alternative way of defining a health impact indicator is used by Goedkoop et al. (1997). They use WHO's concept of DALY, 'disability adjusted life years'. It covers almost all health effects. For each type of health effect the degree of disability or similar is multiplied with its duration resulting in a figure corresponding to "lost person-years".

There is an advantage in using the DALY system in that it gives an overview of all health effects. The reason for not adopting it at present as a default method for the EPS impact indicators is that it includes a large portion of weighting, and that it belongs to the weighting step. Another reason is that a practical degree of resolution when presenting a life cycle assessment in terms of category indicator results would contain about ten indicators. Health effects being perhaps the most important of effects may be expressed in more indicators than the other. A closer look at what types of health effects that are related to environmental issues, we find cancer, respiratory effects, starvation, odour and soiling as common effects. It is regarded desirable to match these types of effects in the category indicators chosen.

However this does not mean that there could not be a harmonisation against the DALY concept.

4.3.2 Production capacity of ecosystems

Decreased yields of crop, fish&meat, wood and freshwater are end point effects associated with production capacity of ecosystems. Different types of crops are grouped together as they may be exchangeable as a source of carbohydrates. Different types of fish&meat may be exchangeable as a protein source. Different types of wood may be exchanged in most applications in a modern society. The indicator chosen for these impact categories is a decreased production capacity of 1 kg. The weight refers to harvest weight for crop and fish&meat, while the dry substance weight is used for wood. Choosing dry weight basis for all three had given the most accurate measure, but normally dry weights are not available for crops or fish&meat while the forest industry often monitors the humidity of the wood it is buying.

The default impact categories and category indicators are summarised in table 4.7 and their compliance with ISO requirement and recommendations evaluated in 4.8 and 4.9 respectively.

Impact category name	**Category indicator name**	**Indicator default unit**	**Notes**
Crop production capacity	Crop production capacity (shortname: crop)	kg	Weight at harvest
Wood production capacity	Wood production capacity (short-name: wood)	kg	Dry weight basis
Fish&meat pro-duction capacity	Fish&meat production capacity (short-name: fish&meat)	kg	Full weight of animals
Base cat-ion capacity	Base cat-ion capacity	H^+ mole equivalent s	Used only when models including the other indicators is not available
Production capacity for water	Production capacity for irrigation water (shortname: irrigation water)	kg	Must be acceptable for irrigation, e.g. with respect to persistent toxic substances
Production capacity for water	Production capacity for drinking water (shortname: drinking water)	kg	Fullfilling WHO criteria on drinking water (1997)

Table 4.7 EPS default impact categories and category indicators for ecosystem production capacity

ISO requirement	How the choice of EPS default ecosystem production capacity indicators comply
The selection of impact categories, category indicators and characterisation models shall be consistent with the goal and scope of the LCA study	Allows weighting and communication
The sources for impact categories, category indicators and characterisation models shall be referenced;	Through this report and the models and data report (Steen, 1999)
The selection of impact categories, category indicators and characterisation models shall be justified	Through this report and the models and data report (Steen, 1999)
Accurate and descriptive names shall be provided for the impact categories and category indicators	Fairly well
The selection of impact categories shall reflect a comprehen-sive set of environmental issues related to the product system being studied, taking the goal and scope into consideration	Includes significant ecosystem production services, but not all

Table 4.8 Compliance of EPS default indicators for ecosystem production capacity with ISO 14042 requirements for the selection of impact categories and category indicators

ISO recommendations	How the choice of EPS default ecosystem production capacity indicators comply
The impact categories, category indicators, and characterisation models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body	Poor compliance at present stage
The impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators	Compliance because indicators are chosen at the endpoint level
Value-choices and assumptions made during the selection of impact categories, category indicators, and characterisation models should be minimised	The intention is followed, but the compliance is difficult to verify
The impact categories, category indicators, and characterisation models should avoid double counting unless required by the goal and scope definition,	Reasonably well met. Some risks of double counting with bio-diversity.
The category indicators should be environmentally relevant	Documented through this report

Table 4.9 Compliance of EPS default indicators for ecosystem production capacity with ISO 14042 recommendations for the selection of impact categories and category indicators

4.3.3 Abiotic stock resource indicators

Natural stock resources

Some authors have suggested impact indicators where several resources are characterised due to their use to reserve ratio or due to their abundance (Lindfors et al., 1994). The reason for this is mainly to be able to highlight the use of scarce resources in an LCA, but it doubtful if this can be regarded as a characterisation based on natural science. It includes a large part of weighting when aggregating resources, many of which find very different use, in one measure. There is no common mechanism like in the formulation of the GWP that applies to the consequences of depletion of different resources.

In the choice of category indicators for the EPS default method, a resource, such as a metal ore is considered unique with regard to the metal. It is only exchangeable with regard to its concentration, chemical composition and location. The impact indicator is therefore defined as 1 kg of the resource in a reference state from which it is mined, i.e. in the state, which is normally referred to as reserves.

Land area is sometimes treated as a resource. Following the second and fifth of the original principles (in table 4.3) for the identification of impact categories and indicators, we find that no change of global surface areas is possible due to human activities. It is only the quality of areas that can be changed. The quality is described by the impact categories: production capacity, bio-diversity and aesthetic values. The qualitative change between land/water may be described by those impact categories.

Anthropogenic stock resources

Buildings, machines, construction materials etc. represent values that can be destroyed by environmental impacts such as corrosion and soiling. There is no separate indicators formulated for these, but the impacts may still be accounted for in the impact modelling. For instance if there is a destruction of steel by corrosion from SO_2, impacts on the safe guard subjects are modelled through just including new pathways in the model. These pathways includes impacts from emissions and resource depletion when substituting and repairing the steel constructions

The default impact categories and category indicators are summarised in table 4.10 and their compliance with ISO requirement and recommendations evaluated in 4.11 and 4.12.

Impact category name	**Category indicator name**	**Indicator default unit**	**Notes**
Depletion of element reserves	= “element name” reserves	kg of element	E.g. Cu reserves, kg Cu
Depletion of fossil reserves	Natural gas reserves	kg	The hydrocarbon part
Depletion of fossil reserves	Oil reserves	kg	
Depletion of fossil reserves	Coal reserves	kg	
Depletion of mineral reserves	= “mineral name” reserves	kg	

Table 4.10 EPS default impact categories and category indicators for abiotic stock resources

ISO requirement	How the choice of EPS default abiotic stock resource indicators comply
The selection of impact categories, category indicators and characterisation models shall be consistent with the goal and scope of the LCA study	Allows weighting and communication
The sources for impact categories, category indicators and characterisation models shall be referenced;	Through this report and the models and data report (Steen, 1999)
The selection of impact categories, category indicators and characterisation models shall be justified	Through this report and the models and data report (Steen, 1999)
Accurate and descriptive names shall be provided for the impact categories and category indicators	The names are descrip-tive and allows reasonably accurate determination of indicator amounts
The selection of impact categories shall reflect a comprehen-sive set of environmental issues related to the product system being studied, taking the goal and scope into consideration	Includes significant abiotic resources, but not all

Table 4.11 Compliance of EPS default indicators for abiotic stock resources with ISO 14042 requirements for the selection of impact categories and category indicators

ISO recommendations	How the choice of EPS default abiotic stock resource indicators comply
The impact categories, category indicators, and characterisation models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body	Weak compliance at present stage, agree by large with documentation from SETAC (Udo deHaas et.al, 1999)
The impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators	Compliance because indicators are chosen at the endpoint level
Value-choices and assumptions made during the selection of impact categories, category indicators, and characterisation models should be minimised	The intention is followed, but the compliance is difficult to verify
The impact categories, category indicators, and characterisation models should avoid double counting unless required by the goal and scope definition, for example when the study includes both human health and carcinogenicity	Reasonably well met.
The category indicators should be environmentally relevant	Documented in this report

Table 4.12 Compliance of EPS default indicators for abiotic stock resources with ISO 14042 recommendations for the selection of impact categories and category indicators

4.3.4 Bio-diversity impact indicators

Our present knowledge about the relation between human activities and decrease of bio-diversity is limited, as is the availability of data on the state of bio-diversity. Some records are kept, e.g. on endangered species (WCMC, 1999) and sometimes the main threat causes. One of the greatest problem in finding suitable impact category indicators lie in the understanding of the quantitative value of bio-diversity. In short term aspects bio-diversity may be considered to be a resource like others, for instance for more efficiently producing food and medicine. In the long run a sufficient bio-diversity is absolutely crucial to human life and there is no trade-off options.

Part of the value of bio-diversity may overlap the safe guard subjects ‘production capacity of ecosystems’, and ‘recreational and cultural values’. The safeguard subject ‘bio-diversity’ is focussing on the genetic resource values.

The most well known change in the safe guard subject bio-diversity caused by human activities is extinction of species. Each year a number of species is extinct. The value given to this change can be estimated from the cost of counteractive measures. Therefore it seems reasonable to use the yearly extinction of species as an indicator for the safe guard subject.

There is however a problem of finding the contribution to the yearly depletion from various activities. One reason is that it is not known exactly which species that are being extinct.

The only way of estimating the contribution to species extinction is via the probability of extinction of red-listed species. Focussing on preserving red-listed species is claimed to be a good strategy for preserving other bio-diversity qualities as well.

So, the category indicator is defined as ‘the normalised extinction of species’ The indicator unit is dimensionless. The category indicator name is shortened to NEX. The normalisation is made with respect to the species extinct during 1990.

Finding an indicator value may be difficult. However, considering that the main threats today are habitat reductions, hunting, harvesting, emissions of toxic substances and similar, estimation of the contribution to extinction may be made on via estimated on reduction of habitat area, number of individuals or number of species in a group or similar.

The default impact categories and category indicators are summarised in table 4.13 and their compliance with ISO requirement and recommendations evaluated in 4.14 and 4.15 respectively.

Impact category name	**Category indicator name**	**Indicator default unit**	**Notes**
Extinction of species	Normalised extinction of species, short-name: NEX	Dimension-less	The normalisation is made with respect to the species extinct during 1990

Table 4.13 EPS default impact categories and category indicators for bio-diversity

ISO requirement	How the choice of EPS bio-diversity indicator comply
The selection of impact categories, category indicators and characterisation models shall be consistent with the goal and scope of the LCA study	Allows weighting and communication
The sources for impact categories, category indicators and characterisation models shall be referenced	Through this report and the models and data report
The selection of impact categories, category indicators and characterisation models shall be justified	Through this report and the models and data report
Accurate and descriptive names shall be provided for the impact categories and category indicators	The names may be difficult to understand directly
The selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied, taking the goal and scope into consideration	Includes a significant aspect on bio-diversity but not all

Table 4.14 Compliance of EPS default indicators for bio-diversity with ISO 14042 requirements for the selection of impact categories and category indicators

ISO recommendations	How the choice of EPS bio-diversity indicator comply
The impact categories, category indicators, and characterisation models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body	Weak compliance at present stage. The focus on red-listed species as a main indicator is international praxis.
The impact categories should represent the aggregated emissions or resource use of the product system on the category endpoint(s) through the category indicators	Compliance because indicators are chosen at the endpoint level
Value-choices and assumptions made during the selection of impact categories, category indicators, and characterisation models should be minimised	The intention is followed, but the compliance is difficult to verify
The impact categories, category indicators, and characterisation models should avoid double counting unless required by the goal and scope definition, for example when the study includes both human health and carcinogenicity	Reasonably well met.
The category indicators should be environmentally relevant	Documented through this report

Table 4.15 Compliance of EPS default indicators for bio-diversity with ISO 14042 recommendations for the selection of impact categories and category indicators

4.3.5 Cultural and recreation value indicators

Changes in cultural and recreational values are difficult to describe by general indicators, as they are highly specific and qualitative in nature. Indicators are therefore defined when needed.

4.4 Assignment of emissions and resources to impact categories (Classification)

When carrying out a specific LCA study, elementary flows registered in the inventory phase are assigned to different impact categories. In traditional LCA, this is a part of the efforts to group and classify elementary flows in order to reduce the amount of impacts that has to be considered. The term classification, as used by SETAC and in the Nordic guidelines (Lindfors et al., 1994) indicates that environmental impacts of the elementary flows are seen as properties of the substances in the elementary flows rather than as a consequence from an exposure situation (you classify a substance). This is also confirmed by the frequent use of the term potential in connections with environmental effects.

In the method described here, a major principle used is to assign an emissions or resource to an impact category when actual effects have occurred or is likely to occur in the environment. This means that an emission is not only defined by the substance in the flow but also by the exposure situation. The default exposure situations accounted for are those, which are present today on a global basis.

The assignment of an emissions or resource to an impact category means not only a recognition that there is a mechanism by which the emissions or other activity is connected to the impact categories, but also that this mechanism is or may be causing impacts in the environment to an extent, which is worth recognising. For instance, any molecule emitted to the atmosphere will adsorb and emit electromagnetic radiation and thus interfering with the global radiation balance. But we chose only to assign those who are known as the main contributors to global warming.

ISO 14042 requests that assigning an emission or resource to an impact category is made on scientific grounds, but the assignment is nevertheless to a large extent a value-laden choice. Which effects should be included, and which effects should be overlooked? The environmental system we define has system borders as well as the technical system, geographical as well as temporal. There are qualitative system borders addressing which types of environmental impacts or changes to include. The LCA practitioner has to decide how to handle impacts that are not subject to scientific consensus. The economy of the LCA study calls for selection of focus. What is essential and what can be neglected?

Consequently, it is suitable to highlight this part of the LCA impact assessment and to report which assignments that has been made and their motives. For the EPS default method, this is made in the models and data report (Steen, 1999).

Emissions to air and water have normally a type of cause-effects chains or pathways that involves dispersion, transformation, deposition and dose-response characteristics. There are however also more direct impacts on the environment, like from physical activities as digging, harvesting etc. These are defined as different types of land use in the EPS default method.

4.5 Characterisation

The characterisation step in a life cycle impact assessment aims at converting and possibly aggregating inventory results to category indicator results. For this it is necessary to find quantitative relations between the inventory parameters and impact category indicators.

An ideal model of a relationship represents a causal relationship. If an extra amount of a substance leaves (or enters) the technical system we would like to find the change in the category indicator occurring in the environmental system. To find this, we have to add up all changes in the impact indicator in all parts of the environmental system that is influenced. This means that dispersion patterns and time constants of the cause-effect chain already become important factors just when selecting system borders.

Many parameters influencing a characterisation factor may be unknown, such as

- future emission and resource depletion volumes. If there is a non linear relationship, the impact from each new human activity will depend on the overall pollution level or resource depletion level.
- spatial information. The location of a source is seldom known. To model an impact we would ideally like to know where the emission or resource depletion take place. For emissions this means also the hight of the chimney or depth in water.
- temporal information. Some products, like buildings, will be used for many years, and we do not know for how long. Some emissions, like NOx, contribute to health effects in a different way during winter and summer, and the contribution have even a diurnal pattern.
- flow volumes. A typical LCA inventory result is expressed in relation to a functional unit. If impact categories are of threshold type, or have other non-linear relations to the inventory parameters, the characterisation factor depends on the flow volume.

Some of the information is lacking because we have no resources to collect it although it exists. Some does not even exist yet and might not be determined even if there were vast resources available.

A common way of dealing with large amounts of information is to use statistical methods. Instead of trying to describe a certain cause-effect chain exclusively, we assign it to a group of cause-effect chains for which the average behaviour is modelled, and we estimate its distribution properties. The final result is then not an exact value, but rather a most probable value or a probability of a certain value being less than a certain level.

Dealing with unknown information on the future, scenario techniques are often used. The most interesting scenario is probably “business as usual”. As LCA most often is used to guide the development of technology in a more sustainable direction, it is meaningful to ask the question: what will be the consequences if we go on like this?
If we examined scenarios where future technology would solve all environmental problems we would not get the incitements to develop such technology.

In order to tell a user of a characterisation factor if the factor is relevant or not, we thus have to make clear for what types of emissions and resource flows it was developed and which impact scenario that is used. The LCA practitioner has to decide which characterisation factors that are most relevant for his/hers purpose. He or she assigns normally inventory flows to a set of “ready-made” characterisation factors representing a certain population of impact scenarios and flows having certain sizes and spatial and temporal extensions.

4.5.1 Modelling characterisation factors.

Characterisation factors express quantitative impacts on category indicators from elementary flows. The size of the impacts often depends on several pathways. Characterisation factors are therefore often a sum of several pathway-specific characterisation factors and each one of these is modelled separately.

There are in principle three types of models used to determine pathway-specific characterisation factors. The first type of methods we may call ‘empirical’, the second ‘equivalency methods’ and the third ‘mechanistic’. The names are chosen because they represent carrying elements in the methods, not because they are purely mechanistic etc.

For the *empirical method*, the system borders in time and space is defined, and the characterisation factor for a substance is determined by dividing a category indicator value allocated to the system with an emission of the substance allocated to the system. The allocation of indicators and emissions may be necessary because of trans-boundary pollution.

Modelling according to the empirical method is most simple if the size of the system is large enough to include all or almost all of the change in indicator value caused by the emission. This happens if the size of the modelled system is larger than $r*u$ and r in space and time, where r is the residence time of the substance in air or water and u its corresponding mean transport velocity. This is also the case if the system borders include all or near all of the target or recipient. Then all effects within the system can be assigned exclusively to the emissions within the system, and all effects caused by emissions in the system will occur within the system.

If some of the effects caused by the emissions modeled are outside the system the allocation becomes more complicated.

Assume we have a system like in figure 4.2.

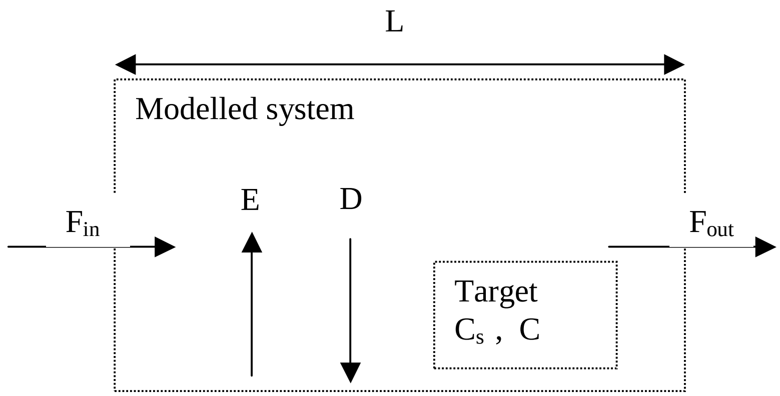

Figure 4.2 Modelling of a characterisation factor with the *empirical method*
F_{in} and F_{out} are the mass transport of the modelled substance in and out of the system, E and D the emission and deposition within the system and C_s and C the system related and total exposure concentrations at the target.

If the total indicator response related to the substance in the system is ψ, the characterisation factor can be written:

$$(f \cdot \psi)/(g \cdot E), \text{ where}$$

f is a factor telling what part of ψ that is allocated to E and g is a factor telling what part of E that should be allocated to ψ.

A way of determining f may be to set it equal to C_s/C. This would be rather uncontroversial if there is a linear dose-response curve. If we have a non-linear dose response curve it becomes more complicated. As there seldom is any information on which part of the dose response curve that is valid at a particular LCA study, C_s/C may also be used as a best estimate for non-linear cases (cf. 3.5.2).

The problem of allocating E is similar to those that occur in the LCI phase, where the modelled system borders are small compared to whole affected system. The solution to the problem is outside the modelled system but there is a lack of information about what is outside. The most simple way of handling this problem is to assume that the outside system is similar to the modelled system and set $g = L/(r \cdot u)$ for $r \cdot u > L$. For air emissions in urban areas however this is a poor model. In these cases the dispersion and target population densities are important and the information about the surrounding system have to be more detailed. In those cases the empirical method become less suitable and we need a more mechanistic approach.

As most characterisation factors, which are used as defaults concern the global system, corrections due to geographical trans-boundary pollution is limited for the initial phase of an EPS analysis. However, the more site-specific the characterisation becomes the more consideration has to be given to this aspect. Sometimes, as in the case of CO_2, there is a trans-boundary pollution in time.

The *equivalency method* use traditional equivalency factors like the global warming potential to calculate characterisation factors. For example to calculate CO's impacts on severe morbidity via global warming, the characterisation factor for CO_2 with respect to severe morbidity is used and just multiplied by the GWP for CO relative CO_2.

The *mechanistic method* typically estimates the portion of an emitted amount of a substance that will reach a sensitive target and use dose-response information to calculate the response per mass of the emitted substance.

4.6 Weighting

The term 'weighting' came to substitute the term 'valuation' during the development of ISO 14042. A major reason for that was to find a broader consensus in a term that did not emphasise the subjective element in this LCA step. There are many methods that allow comparison across impact categories, where the subjective element is limited to the choice of the weighting principle, like for instance the MIPS-measure (Schmidt-Bleek, 1994) where the total mass flow is used as an overall measure.

However in the EPS default method, the weighting is still made through valuation.

Although not explicitly expressed in ISO 14042, weighting requires definition of weighting indicators and weighting factors in a similar way as for characterisation of emissions. For the EPS default method there is only one weighting indicator, as only one value for the total environmental impact is requested.

4.6.1 Definition of default weighting indicator

As discussed in 3.5.1, the default-weighting indicator, which is preferred, is the willingness to pay (WTP) to restore impacts on the safeguard subjects, as measured amongst today's OECD inhabitants. The choice of today's OECD inhabitants is made in order to facilitate the understanding by the designer, who most likely is an OECD inhabitant or a person outside OECD with good contact with the OECD world. Today the OECD countries have a dominating role in the development of new technique and are beginning to adopt the ideas of sustainable development. Of course there are many other cultures that can claim to be more sustainable than those of the modern OECD countries, but their limited use of tools like LCA makes it more reasonable to investigate the consequences of their attitudes as options and not as a default.

The choice of default reference state is the environment of today. The reasons are similar as for the choice of WTP. There is a need for an understanding of what the reference state look like. Today's situation is real to us and can easier be communicated than a hypothetical state like 'the untouched nature'.

4.6.2 Methods to determine default weighting factors.

Weighting factors are the ratio of weighting indicators and impact category indicators. They represent the WTP for one indicator unit. They are separately modelled and a set of models (factors) is used 'ready-made' by the LCA practitioner.

WTP for category indicator units may be estimated by various methods. Various methods tend to give different results. However this is not a serious problem, and may be addressed in the same way as measuring emissions. The uncertainty is estimated and expressed as a distribution function.

For some category indicators, the market price may be used to estimate WTP. It may be disputed whether the marked price is what is paid or if various subsidiaries and taxes should be included. For instance, what WTP should be used for crop? Is the world market price that is paid directly better to use than the price the society pays, which mainly is the sum of the buyers costs and the cost for subsidiaries. If we accept to add the cost of subsidiaries we also have to accept the subtraction of taxes if we want to be consequent.

However, the goal that was set up for the EPS system requires the result to be understandable for the designer. This speaks for a choice of a monetary value that is familiar to the designer: the price the buyer has to pay.

When studying the market prices you find great variations, partly because there are differences between different regions but also because the category indicators are not sharply defined. Crops includes all sorts of crops, like oat, wheat, barley, rice and corn, and their prices vary on the market. These variations are included in the uncertainty measure of the weighting factor.

If there is no direct market, where the indicator value may be found, there are several other methods used for finding the WTP. Some involves studies of behaviour, like the hedonic pricing method, where estate prices are used or like studies of travelling. Both use the extra costs taken to reach a better environment as a measure of the WTP.

A method often used to estimate non-market environmental values is the CVM method. CVM stands for 'Contingent Valuation Method' and is widely used to measure WTP in various groups to various concepts, which are described to them. The CVM technique is based on interviews and is following a special procedure. In the EPS-system the CVM technique is used for category indicators of morbidity and nuisance and for recreation values. The precision of the CVM technique varies.

When trying to find the WTP for indicators of the safe guard subject ‘abiotic stock resources', we find that neither the market nor the customers are available to study. You cannot use the CVM technique to determine the WTP for those that are concerned, because most of them belong to future generations. There is no one to ask. To cope with this in the EPS default method, a market scenario was created, where the production cost of substances similar to the abiotic stock resources is used as an estimate of WTP. It is assumed that some of these stock resource materials always will be produced even if the volumes decrease. Consequently there is a will to pay at least it takes, but probably, in the long run, not much more.

4.7 Analysis of sensitivity and uncertainty

As indicated in 3.4 life cycle impact assessment involves large uncertainties. Some of these come from known variations in data, some are of methodological and epistemological character.

It is vital to know how these uncertainties influence our conclusions. In the EPS system there are methods developed for analysis of sensitivity and uncertainty, using estimates of identified uncertainty for individual input data.

Uncertainties due to deficient knowledge of mechanisms and processes are however not dealt with.

Life cycle assessments are normally made without quantitative estimations of accuracy or precision. In SETAC´s ‘Code of practice’ (1993) sensitivity and uncertainty analysis are recommended, but the methodology is not very well developed. In the ISO 14040 sensitivity analysis is requested.

Hoffman et al.(1994) reviewed statistical analysis and uncertainties in relation to LCA discussing technical, methodological and epistemological uncertainty (e.g. from lack of knowledge of system behaviour). Heijungs (1997) developed a sort of sensitivity analysis called ‘dominance analysis’, where the most important contributions to the result are identified. Kennedy et al.(1996) makes an uncertainty analysis of the inventory part of an LCA using beta-distributions.

4.7.1 Terms and definitions

The term *significance* is used in a traditional meaning based on the probability of a statement being true, for instance: one concept being better than another or that a single concept leads to an impact less than a given level of ambition.

The term *sensitivity* may sometimes be unclear. One may ask the question: sensitivity of what for what? Is it how a single parameter depend on another in a continuos function, is it how the resulting impact quantity depend on allocation rules or system borders or is it how sensitive a priority obtained is to uncertainties in input data?

For a continuos function $y = f(x)$ the sensitivity is normally defined by a derivative $\partial y/\partial x$. y can be thought of as the result and x as any input parameter. The parameter y changes with x according to the derivative. In linear expressions, which are normally used in LCA, the derivative is constant.

A non-linear function may exist when an input parameter, x, represents an emission causing an effect where there is a threshold. The sensitivity may then be defined for ranges of x's, but this is probably only meaningful when there are few such ranges or independent non-linear parameters. If the sensitivity expression becomes too complex, it looses its informative capability.

When changes in allocation rules or system borders are made there is a discrete discontinuous change in y. The sensitivity is then a change in the parameter y as a result of a specific change in an allocation or a system border.

A particular type of sensitivity is at hand in a comparative LCA. Is A better than B or vice versa? The sensitivity of the priority given to changes in the input parameters expressed as a derivative is not very informative as it is always zero as long as the derivative exists. There is no change in the result (priority) when the parameter changes until we reach the transition point, where there is a discontinuity and a derivative cannot exist. In the EPS default method the sensitivity of the priority to a given parameter x is expressed by means of the change Δx, necessary to change the priority obtained in a comparative LCA. This number itself may however not be very suitable to use as a measure of sensitivity as it increases when the sensitivity of the priority to a parameter decrease. It is therefore referred to as *the critical error* in x and $\Delta x/x$ as the *critical error factor (CEF).*

The sensitivity of the priority to various input parameters is easier to understand and remember when using a *relative sensitivity*, $\sigma_x/\Delta x$, where σ_x is the uncertainty of x expressed as a standard deviation. The ratio tells us how important the precision in the estimation of x is for the priority obtained.

When making an improvement on the basis of a weighted LCA result, uncertainty in the input data leads to a certain probability of making the wrong decision. In those cases, the impact on the environment is negative. To describe the consequences of uncertain input data on improvement of the environment we define a term *net improvement efficiency* which is equal to the ratio of the average environmental impact improvement to the highest possible improvement, which is obtained if all decisions are correct.

4.7.2 The log-normal distribution

As uncertainty and variability in data often is large and as a large number of factors influence the data, the log-normal distribution is mostly relevant to use. Even if variations are small and the ordinary normal distribution can be used the log-normal distribution is preferred in this work unless there are special reasons. Natural logarithms are used.

If an uncertainty interval is identified, where almost all variability may be included or within which the true value lies with a high degree of probability, this is taken as 2 standard deviations from the average of the log values.

In a log-normal distribution the average of the log values is not the same as the average of the original values. But the medians are the same.

If we want to find the original value corresponding to plus one standard deviation in the log values it may be found by multiplying the original median value (M) by a factor being the anti-log of the standard deviation in the log values. A standard deviation in a log-normal distribution of 0.6931 thus corresponds to an uncertainty of a factor of two in the original values. $\exp(\ln M + 0.6931) = \exp(\ln M + \ln 2) = \exp(\ln(M*2)) = 2M$.
Or more general: $\exp(\ln M + \sigma) = \exp(\ln M + \ln(\exp\sigma)) = \exp(\ln(M*(\exp\sigma))) = M*(\exp\sigma)$

To translate the interval of 2 standard deviations in the log values to original values we have to multiply the median with the square of the factor corresponding to one standard deviation. $\exp(\ln M + 2\sigma) = \exp(\ln M + \ln((\exp\sigma)^2)) = \exp(\ln(M*\ln(\exp\sigma)^2)) = M*(\exp\sigma)^2$.

As it is easier to conceptualise the original values, values of σ are seldom mentioned. Instead values of σ, is referred to as a 'standard deviations corresponding to a factor of expσ'.

When modelling characterisation factors, the result is mostly average values. To find the corresponding median values to be used in distribution modelling and in Monte-Carlo simulations to determine the overall uncertainty, a numerical PC program has been used. In this:

1. an initial value of the median is guessed,
2. the average of the original values is calculated and compared to the real one
3. depending on the comparison the guessed median value is increased or decreased with an increment.
4. A new average of the original values is calculated and compared to the real one
5. Etc. , until the average values are equal

Standard deviation or uncertainty intervals may sometimes be derived from earlier estimations. For instance if a characterisation factor is determined by multiplying two factors x and y, each having a log-normal distributed uncertainty and which logged standard deviations are known, σ_x and σ_y, the resulting standard deviation will be $\sqrt{\sigma_x{}^2 + \sigma_y{}^2}$ or if expressed as 'corresponding factors' in original values the new factor will be $\exp(\sqrt{(\ln(\exp\sigma_x))^2 + (\ln(\exp\sigma_y))^2})$.

4.7.3 Uncertainty in input data

A life cycle assessment contains a lot of input and output data. The input data are normally average values representing a population of data or some times single values. All data are uncertain to some extent. One may describe this uncertainty as a probability distribution. The probability distribution may be a normal or log-normal distribution or be characterised by some other expression.

Sometimes an allocation rule or a choice of a system border causes the uncertainty. In such cases one may analyse the consequences of alternative choices one by one, or chose a probability number for each alternative choice.

There are at least two types of uncertainty involved. One is the normal uncertainty associated with the determination of a parameter in a given system. The other is associated with the choice of such a parameter value to represent a value in another similar system. If we are going to use a truck for transportation of a future product, we normally use emission data for trucks representing some average driving cycle and load. Such data are typically some years old. We thus use old data for a given situation to represent a future situation with different driving conditions. An extra error or uncertainty is added. This means that a user of data from a database would have to reconsider uncertainties given in the database if the data populations are not the same.

The use of linear relations instead of more accurate non-linear relations introduces an extra uncertainty.

It is often argued that the estimation of uncertainty in itself is very uncertain in LCA studies. This may explain the striking lack of such estimations in reports and publications.

Even if a strict calculation of distribution parameters in a random sample of the parameter x is not available, it may still be meaningful to give a number to the magnitude of for instance the standard deviation of the distribution. Considering the very large difference in precision between estimations of economically important parameters, for instance oil consumption, and more peripheral parameters such as trace metal concentrations, which are measured only occasionally, it seems likely that this information may add something to the life cycle assessment.

Variations in series of measurements of emissions and process parameters, comparative studies with different methods, and analysis of models used to estimate characterisation factors are sources of information that may be used to estimate distribution parameters.

After having made a full sensitivity analysis the sensitivity to errors in error estimates may also be analysed.

4.7.4 Uncertainty in output data

As input data used in calculations may have different distributions, and as an LCA normally involves solving large matrixes it is hardly practical to formulate an analytical expression for the probability distribution of the calculated values unless the calculations made in the LCA are very simple. A more general and convenient way is to use numeric simulations. Most of the calculations of the LCA are made by computer programs and the extra programming necessary to make numeric simulations is limited.

In the spreadsheet program Microsoft® Excel, random number generators are available for even probability distributions and for normal distributions.

It is however important to remember that some of the input parameters are not independent. For instance, in a comparative LCA where both concepts use transports with emissions that have been estimated by means of the same emission factors the consequence of a too large value will be an increased environmental impact for both concepts. Therefore the LCA calculations should as far as possible be made "from the cradle", which means that a primary value is entered only once in a calculation model. An example will be given below.

Assume that the calculated LCA value for a design concept is y, determined by the expression

$$y = i_1 \cdot c_1 \cdot v_1 + i_2 \cdot c_2 \cdot v_2 \qquad \text{............(1)}$$

where i_1 is the inventory result of emission 1, c_1 the characterisation factor of emission 1, and v_1 the valuation factor for the characterised emission 1 etc.

Assume that the parameters of equation (1) have values and distributions according to table 4.16 below.

Parameter	Best estimate	Distribution function	Distribution parameter type	Distribution parameter value
i_1	2	Log-normal	standard deviation, (σ_x)	1.2
c_1	1	Log-normal	standard deviation, (σ_x)	2
v_1	5	Log-normal	standard deviation, (σ_x)	3
i_2	3	Log-normal	standard deviation, (σ_x)	1.5
c_2	2	Log-normal	standard deviation, (σ_x)	2.5
v_2	1	Log-normal	standard deviation, (σ_x)	5

Table 4.16 Values and distributions in demonstration example of equation (1). All distribution functions and distribution parameter types are the same in this example but they could as well be of different kinds.

The values are fictive examples but not unrealistic. Inventory data for emissions of NO_x and SO_x are often determined by an overall accuracy of about 20% and characterisation factors vary typically with and factors of 2-3 for different circumstances (Lindfors et al., 1994). Valuations as determined in western populations vary even more (Strand, 1991).

The value of y in this example is 16 if no uncertainties are included in the analysis.

If small errors are generated to give a log-normal distribution of the input parameters we obtain values similar to the best estimates but being somewhat larger or less. For instance for i_1, with the best estimate of 2, values like 1.80266, 2.143637, 2.345679, 1.68151...etc. would be obtained. In a simulation with 100 calculations and random errors on all input parameters y got values as shown by figure 4.3.

Normally 100 simulations seem to give enough information on the precision of the results but if a reproducibility on the percent level is wanted 1000 simulations is more relevant.

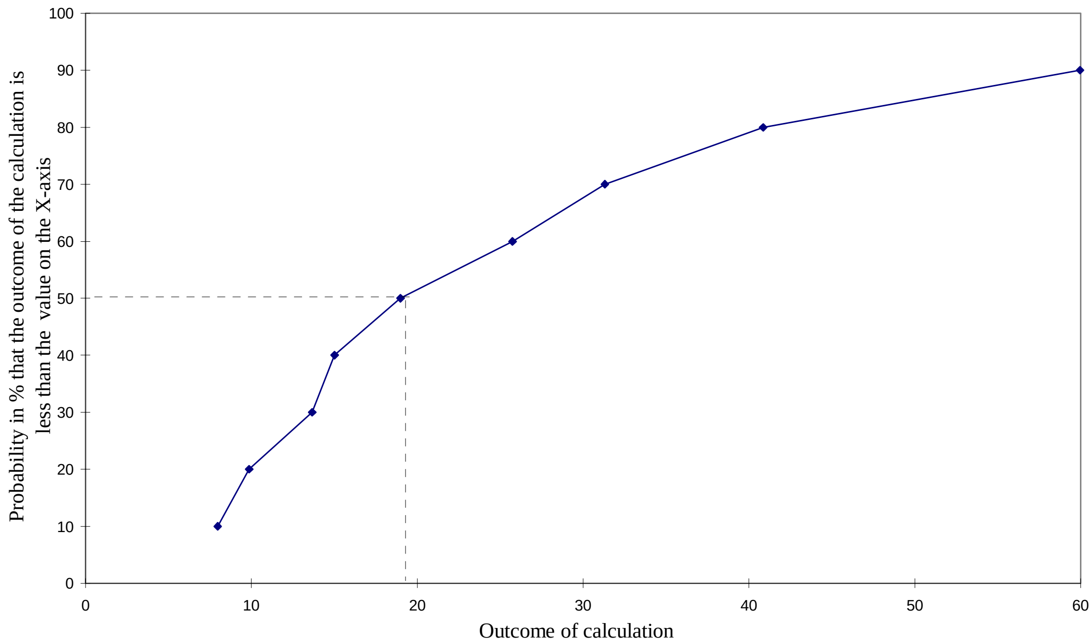

Figure 4.3 Cumulative distribution of results from equation (1) when random errors are added to input data for i, c and v. Note that the median value differs from the first estimate, 16. This may be explained by the skewed log-normal distributions, which means that the value of y will increase more when a parameter value is above median than it will decrease when it is below.

The result above may seem rather depressing as we are used to have better precision in most quantitative calculations. However, the result y is very seldom used for decisions. It is more common to have a comparison between two concepts, and the result y_A - y_B is of more interest. Or rather the sign of the expression $y_A - y_B$. If concept A is better that B, which means it has a smaller y value, $y_A - y_B$ is negative.

If we thus let table 4.16 above represent concept A and table 4.17 below represent concept B, where B emits the same types of substances, we get the same characterisation and valuation factors, but different inventory data.

Parameter	**Best estimate**	**Distribution function**	**Distribution parameter type**	**Distribution parameter value**
i_1	3	Log-normal	standard deviation, (σ_x)	1.2
c_1	1	Log-normal	standard deviation, (σ_x)	2
v_1	5	Log-normal	standard deviation, (σ_x)	3
i_2	2,5	Log-normal	standard deviation, (σ_x)	1.5
c_2	2	Log-normal	standard deviation, (σ_x)	2.5
v_2	1	Log-normal	standard deviation, (σ_x)	5

Table 4.17 Input parameters to equation (1) for concept B.

The best estimate for y_B is 20, thus giving $y_A - y_B$ equal to -4.

If we in the same way as for the calculation of y_A add random errors to all “i”, “v” and “c”:s all “i”:s would get different random errors, while c_1 is the same random error for A and B. This means that if c_1 gets a too low value, both y_A and y_B becomes too low and the change in $y_A - y_B$ is less than if c_1 for A and B each varied independently at random.

A cumulative distribution of the results from 100 calculations of $y_A - y_B$ is shown in figure 4.4.

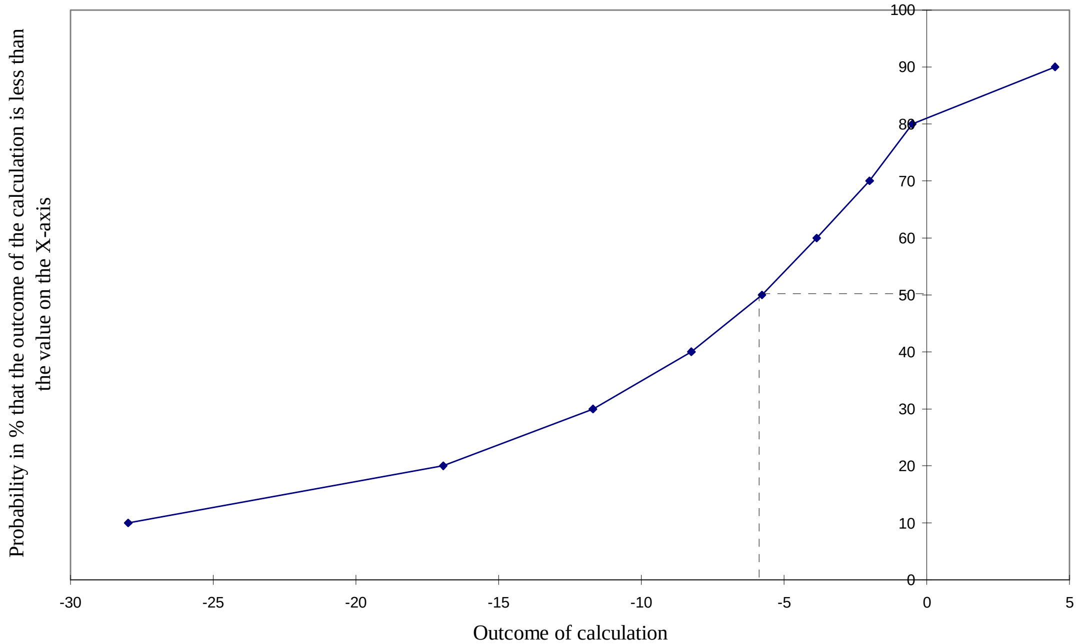

Figure 4.4 Cumulative distribution of y_A - y_B values when random errors are added to input data for i, c and v.

The results as shown in figure 4.4 tell us that the probability of concept A being better than concept B is about 80%.

There is however more information for a decision-maker in figure 4.4.

If concept A is chosen before concept B, the probability of decreasing the environmental impact is 80% and the probability of increasing it is 20%. If many decisions are made due to results like in figure 4.3, the net improvement efficiency would be approximately 80 - 20 = 60%. More precisely the net improvement efficiency of choosing A would be $(T_2-T_1)/(T_1+T_2)$,

where T_1 and T_2 is the surfaces between the curve and the Y-axis at the interval 0-100 above and below the point of interception respectively. T_2-T_1 represents the net improvement when making some right and some wrong decisions and T_1+T_2 represent the maximum achievable improvement when making the right decision every time.

The improvement efficiency may be increased either by improving the database for the calculations or by changing the technical concept so that the marginal becomes greater for the best performing alternative.

In a way one could say that the value of data quality is reflected in the cost for increased marginal of environmental performance necessary for new technical concepts to give significant improvements.

4.7.5 Sensitivity analysis

<u>Sensitivity of aggregated data to input data</u>

In the example above in table 1, $y = i_1 \cdot c_1 \cdot v_1 + i_2 \cdot c_2 \cdot v_2$, and all sensitivities of y_A to changes in input parameters may easily be calculated. For instance $\partial y_A / \partial i_1 = 5$ as $c_1=1$ and $v_1=5$.

But what does the sensitivity figure mean? One may of course find out for which data y_A is most sensitive. Heijungs (1997) has developed a methodology for this. But is the uncertainty acceptable or not?

It is not until data is used in a comparison or decision, this question may be answered. Therefore the sensitivity of priorities will be addressed below.

<u>Sensitivity of priorities to input data</u>

Through the results presented in figure 2, we found that the significance in the statement that concept A was better than B was about 80%.

A sensitivity analysis may tell us what is contributing most to the low significance. The relative sensitivity for the example above is calculated and the results shown in table 4.18.

parameter	**operation to make y_A-y_B = 0**	**critical error factor (CEF)**	**error factor**	**relative sensitivity**
i_{1A}	multiply by CEF	1,4	1,2	0,857
i_{2A}	multiply by CEF	1,667	1,5	0,800
i_{1B}	divide by CEF	1,363	1,2	0,880
i_{2B}	divide by CEF	5	1,5	0,3
c_1	divide by CEF	5	2	0,4
c_2	multiply by CEF	5	2	0,4
v_1	multiply by CEF	5	3	0,6
v_2	multiply by CEF	5	5	1

Table 4.18 Calculation of relative sensitivity.

From the results in table 3 it can be seen that the largest relative sensitivity of the priority is to v_2, which means that it has the highest contribution to the uncertainty in determining the priority between concept A and B. The inventory data i_{1A} , i_{2A} and i_{1B} has a relative sensitivity that is close to 1 and is also contributing to a low significance of the results. So in order to improve the LCA study the data v_2 , i_{1A} , i_{2A} and i_{1B} are the ones that should be considered first.

In a more generalised way we can write any aggregated LCA result:

$$\sum i_j \cdot k_{jk} \cdot v_k$$

,where i_j is the j:th inventory result, k_{jk} the characterisation factor between inventory parameter j and impact indicator k, and v_k is the weighting factor for impact indicator k.

Any change of a parameter *i*, *k* or *v* will thus result in a linear response of the aggregated result (figure 4.5). If , for instance the parameter is a characterisation factor and the corresponding inventory parameter values in concept A is less than in B. the slope is less and the priority will change if the characterisation factor value increase to a certain level.

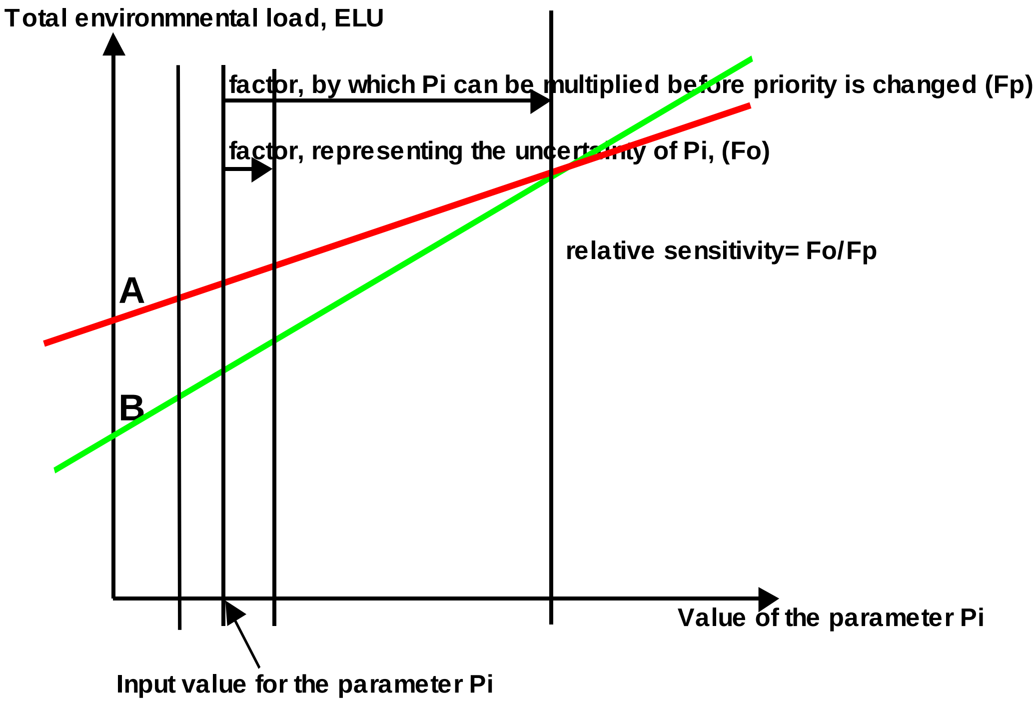

Figure 4.5 The total environmental load is linearly dependent on all input data

5 Optional methods

The EPS system is designed to allow alternative impact assessments when needed. For instance, when a supply chain has been identified and the particular locations of important production plants are known, more specific temporal and spatial conditions may be entered into the models used for characterisation factors.

Another reason for using alternative impact assessments could be to implement a company's environmental policy, which may give other priorities than what is obtained by the default method.

Using several optional methods is a way of getting a more comprehensive view of various aspects of the added impact from a product and alternative product concepts.

Lindfors et al.(1994) recommend that several weighting method is used at a life cycle impact assessment, but it may be difficult at an early stage in a design process.

5.1 Changing system borders and scenarios in default method

5.1.1 Temporal system borders

If the temporal system borders are decreased down to a few hundred years, a significant decrease in the weighting of abiotic stock resources occur. A time scale is then approached for which many of the present reserves or anticipated future reserves may not be depleted.

When decreasing the temporal system borders to less than 100 years the type of effects that has to do with decreased mineralisation from toxic metals in soil tend to be important. Emissions of arsenic, copper, cadmium, chromium etc. gets significantly increased weights, as only the initial toxic effect on micro-organisms and the subsequent decrease of available nutrients is included and not the compensating increase that occur after 100 years.

Another significant change when decreasing the temporal system borders is the decrease of the weights for greenhouse gases. Much of the effects are on the 100 years scale. If temporal system borders are less than 100 years, part of the effects will not be included.

Instead of a distinct cut off for the temporal system borders, a shift in focus may be achieved by using discounting.

5.1.2 Spatial system borders

In the default method average impacts on the globe from average emissions on the globe are considered. The precision in the impact assessment may be increased considerably if the location of a source is known. Then the spatial system borders for the emissions and for some of its effects may be much narrower. One reason for narrowing the spatial system borders for the effect is that it may only occur locally and consequently the modelling become easier. Another reason could be that management of local effect is given priority compared to management of global effects.

5.1.3 Technological and societal scenarios

In order to be able to estimate future impacts assumptions have been made on which society and available technology there will be. The scenarios used in the default method are of 'business as usual'type and a very conservative approach is made to future technological progress. The scenario setting in the default method is particularly important for long-term effects like the greenhouse effect, the depletion of reserves and for bio-diversity.

If, for instance, it were assumed that economic growth in Africa increased considerably, the consequences of global warming would be much less. Then, importing from other regions could compensate a decrease in food production.

Some scenarios used in the default method describe the sustainable production of abiotic reserves. It would not be unrealistic to assume that some of the reserves may be substituted in another way. For instance, micro-organisms may gather some of the very scarce metals or plants and some metals may be produced from higher grades than assumed in the default method.

5.1.4 Environmental scenarios

If the temporal and spatial system borders change, there will be another type of environment to include in the modelling of characterisation factors.

There are also optional scenarios that may be of interest when keeping the system borders as they are in the default method. For instance, dose-response curves for the sensitivity of crops to air pollution may change due to the introduction of more resistant types. Or, counteractive measures, like setting aside protected areas, may reduce the vulnerability of red-listed species for forestry and agriculture.

Some of the models for effects caused by global warming are very uncertain, and alternative scenarios may be formulated, for instance on the impact on bio-diversity

5.1.5 Scenarios with alternative value-settings

It may be of interest to examine the outcome of weighting factors that represent attitudes of other cultures or anticipated changes of life conditions in the near future for the OECD population. There has been a tendency in the last decades of increasing the value of human health compared to the value of food. This tendency could continue, but it could also be the reverse.

Today the concern of bio-diversity is most likely based on a scenario that we after all may manage to keep it on a ‘functional’ level. If signs occur that this may not be the case, a significant change may be expected in the weight given to the threat of red-listed species.

5.2 Using alternative indicators and weighting methods

In 3.5, alternative weighting methods were discussed. Most of them can be adapted to the EPS system. There is however a need to make additional estimates on uncertainties of the characterisation and weighting factors.

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